Master of Forestry Science

Carbon sequestered by native restoration plantings, southern Port Hills and Quail Island, Canterbury.

Student: Paula D. Yarur Thys

Student ID: 29574350

Supervisor: David Norton

Co-Supervisor: Euan Mason

April 30th 2021

School of Forestry, University of Canterbury

Abstract

Restoration plantings are typically planned with a multi-goal framework. Carbon sequestration has been a common component of these since the early nineties, but we know little about how plantings in different locations achieve this goal and what improvements can be made. This study investigated the above-ground biomass (AGB) held at five native restoration sites on the southern Port Hills and Quail Island. Date of planting was key to locate randomised sampling sites in satellite imagery using GIS. Temporary plots were set out in the field to measure all AGB and two allometric equations were applied to the measured data to estimate carbon expressed as CO2 equivalents. Data analysis was undertaken using R statistical software. A comparison was made between the primary allometric equation used in New Zealand's national carbon accounting system (which is based mainly on mature trees) and an allometric equation based on shrubs. Results showed an average 9% increase in the total CO_2 equivalent for all plots when using the mature tree equation for plantings less than 60 years old. A revision of the allometric equations used may improve the accuracy of the carbon accounting system if time and costs are not a limitation. Plot species-composition variables and environmental variables were not found to have a significant influence on CO_2 equivalent in the restoration plantings. However, the CO_2 equivalent amounts differed from those of the MPI native species look-up tables at 30 years post planting. The values suggest that a restoration planting with biodiversity objectives can reach higher carbon content the older they are and up to at least 59 years with no indication of an abatement in this. It was also found that *Podocarpus totara* can be used as an enhancement species to increase CO_2 equivalent levels in restored areas. To enhance CO_2 equivalent levels in tōtara plantings, proper management is necessary including elimination or at least reduction of ungulates and for trees to be planted on the forest's edge. Attention to management, species composition, and use of enrichment species for current restorations would improve carbon content and hence to efficiently achieve our carbon goals.

Table of Contents

Table of Figures

Table of Figures

CHAPTER ONE – Introduction, Background and Study Aims

Introduction

The UN declared 2021-2030 the decade of ecosystem restoration, as it acknowledges the need to restore damaged ecosystems to secure a sustainable future (UN, 2019). New and restored ecosystems will become crucial during the next few decades in the face of global climate change. It has been suggested that under several monetary and policy assumptions having healthy ecosystems such as forests acting as carbon sinks will mean one-third of global CO_2 emissions can be mitigated between 2017 and 2030 to achieve a less than 2°C temperature increase (Griscom et al., 2017). Restoring ecosystems provides numerous benefits to a society as well addressing carbon emissions. In New Zealand, the Climate Change Commission draft report identified an increase of native forests for carbon removal but did not differentiate between regenerating native forests and restoration plantings (Commission, 2021) and both are likely to be important. Recreation, aesthetics, conservation and biodiversity are usually the main objectives of restoration plantings, and few studies have investigated the relationship with carbon content, making it more difficult to understand the difference between these plantings and natural regenerating forests. Further research can allow us to understand how variable the carbon content in restoration plantings is and if a dominant canopy species will enrich these areas by providing higher carbon content to existing and future plantings.

New Zealand's landscape has gained patches of restored indigenous forest through plantings achieved through community efforts (Norton et al., 2018). These plantings are mainly volunteer-based with modest funding which constrains the levels of maintenance and the land area it is possible to cover (cf. plantation forest). These restored ecosystems are scattered around the country, and we know little about their carbon contribution and management for enhancing it. Many of these plantings are eligible for carbon credits, an economic stream to help increase our restoration projects. These credits are based on a general carbon look-up table for all indigenous forest species around New Zealand, including different environments and management. Much like the Climate Change Commission report, a general table encompassing a vast variability that assumes every regenerating native forest sequesters the same amount of carbon, accuracy to contribute an adequate monetary value to an existing carbon content is unrealistic. There is a need for knowledge around the current restoration efforts we have in our landscapes to improve their future carbon content and carbon removal. Such knowledge will help us apply a genuine monetary value to the different sites, species and management of our future ecosystems.

Measuring live, above-ground biomass in the local indigenous planted forest of the southern Port Hills and Quail Island will provide local data to help provide locally-relevant carbon sequestration values for better management and planning practice. The lack of a differentiation in the carbon sequestration of restoration plantings is because few studies are carried out during the early stages of restoration plantings of mixed native secondary growth. Most of the carbon sequestration data is based on single-species plantings or on naturally regenerating shrublands that grew without planting (Carswell et al., 2014; Carswell et al., 2009; Mason et al., 2014). The importance of understanding the carbon content of native plantings, comprising mixed short- and long-lived species, is that they sequester carbon for longer periods and are excluded from harvest continuously increasing their biodiversity and carbon values. Because they are perpetual forests that reintroduce native species, the carbon sequestered becomes a permanent carbon store and measuring the uptake in early years will facilitate understanding appropriate management requirements and carbon outcomes. The site characteristics, management and composition of restoration plantings are typically highly variable, therefore the amounts of carbon content are bound to vary too. By better understanding their carbon content and the factors that influence it, we will understand what carbon amounts we can obtain from these efforts and if any improvements can be done to current restoration management practices.

Background

There is increasing interest in the ecosystem services and other benefits, such as carbon sequestration, that trees provide (Shields et al., 2016). While there are multiple ways we can achieve these benefits, a strong focus in New Zealand in the last few years has been around the idea of planting the right trees in the right place with a focus on native species (Monge et al., 2018; Scion, 2018). New projects such as the One Billion Trees programme have promoted a change in the landscape by encouraging communities and landowners to plant more trees. An initiative like this will introduce change to the landscape and consequently, we will see more plantings with carbon sequestration as a goal. If we are trying to plant more trees, we must focus our efforts in the right direction. Planting trees requires a broad base of information to support the decisions to choose the most suitable species. Maintaining and enriching these plantings are also crucial parts of ecosystem restoration and effective carbon sequestration (Forbes et al., 2020). Having evidence of the carbon content in biomass of the variety of plantings available, while including their range of variation, will help us quantify carbon contents in community plantings. These are important plantings as they are happening in many parts of the country. With over 600 environmental community groups in New Zealand, 380 of which manage forest ecosystems, this is not a minor action as most of these groups are over 11 years old (Peters et al., 2015). To better represent the carbon uptake from these plantings, studying the sequestration they have achieved is important. The more knowledge we have about species carbon sequestration across a range of sites, the more we can optimise our restoration efforts and results.

Carbon and climate change

The New Zealand Government's approach to addressing climate change includes the target of becoming carbon zero by 2050. The use of carbon credits has been a provisional approach to offsetting emissions and earn time before sustainable long-term solutions to reducing emissions are developed (Leining & Kerr, 2018). Forest carbon sequestration has been the critical tool, but we must be careful as it is unwise to keep assuming that carbon sequestration of native forests is the same for all species and conditions.

The Ministry of Agriculture and Forestry (MAF), a historic government agency whose services now fall under the Ministry for Primary Industries (MPI), developed a single table to describe the sequestration rates of all native forests (MAF, 2009). MAF initially set the carbon sequestration value for indigenous forests as $3tCO_2e$/ha/yr, which was later amended following further research. This research was undertaken by Landcare Research and Scion and was based on regenerating indigenous shrublands. From these studies only one written report is available which clarifies the methodology, sites, and species composition (Payton et al., 2009) and was used to derive the current look-up table used by

MPI. The look-up tables are the default method to calculate carbon sequestered by forests so that farmers, landowners, and restoration managers can calculate carbon credits and register their forest through the Emissions Trading Scheme (ETS), regardless of their management. Through this process, the ETS provides landowners with an additional income stream from the carbon sequestered by their plantings as a monetary return for this ecosystem service. Landowners are required to comply with the regulations of the scheme and maintain the forests through time to secure their forests as future carbon sinks. If the forests are removed or damaged the ETS has a system for landowners for reporting and repaying the carbon content that is lost in such events (Acosta et al., 2020). The importance of providing the best representation of the carbon sequestration from restoration plantings is high as it impacts directly on the economic sustainability of restoration efforts, and therefore on achieving our climate change goals. If we help sustain the budget of current restoration projects we are more likely to achieve the goal of protecting them as future permanent carbon sinks, a valuable approach to help sequester our currently exceeded carbon emissions (Holdaway et al., 2017).

Restoration plantings are complex biological systems, primarily because they are not plantations of a single species such as the case of exotic forestry, nor do they have the same species in every planting. Because there is a range of different species used, we must understand the carbon sequestration rates to represent distinct mixes of species-specific plantings if we are to offset real amounts of carbon sequestered through carbon credits. The lack of detailed studies makes it difficult to understand how much carbon is being sequestered. With more plantings being introduced into our landscape to reach the 2050 goal of becoming carbon zero, knowing what has already been captured in local restoration plantings will be of great value for future carbon sequestration rates applied to these new plantings.

Another important consideration in comparing exotic plantations and restoration plantings is their species composition. Exotic fast-growing trees sequester carbon rapidly, but their biodiversity and cultural values are much less than those of indigenous plantings. Harvesting trees for wood at the same time as using these for carbon credits has been common and exotic plantations are also assumed to be carbon neutral when they reach maturity as they do not sequester any more additional carbon once they have been harvested as they are continuously replaced when reaching the same age (Asante & Armstrong, 2012). If we think about it, this is counter-productive because at the moment of harvest, we release the sequestered carbon faster than a new seedling will be able to absorb it back when the replanting happens, plus the wood product industry has associated carbon emissions that need to be accounted for (Ford-Robertson, 1996; Gepp & Wright, 2019). It is not wrong to have both objectives, but it is controversial and counter-productive for the carbon sequestration goal as at the moment of harvest they are no longer effectively mitigating carbon emissions that are already in the atmosphere (at that point in time). Native restoration plantings that are not harvested (which currently is the intent of all restoration plantings) provide a more long-term and sustainable solution for carbon sequestration while also providing important native biodiversity and cultural values.

Choosing the right tree to achieve a combination of different planting objectives can be most effective if based on the local situation. Not all trees will sequester the same amounts of carbon, nor will they do so at the same rate. Species selection for carbon purposes is important, and usually it will not be a single objective planting. With native species plantings, biodiversity conservation is usually the main objective, with carbon sequestration being secondary. However, it may be that having more permanent native forests in the landscape means locking in a future carbon store that doesn't disappear and can be done on relatively unproductive land. Carbon sequestration as an objective can be combined with other purposes if plantings take place in erosion-prone areas and in areas important for recreation such as the southern Port Hills and Quail Island. These are both places where biodiversity and carbon sequestration can benefit stakeholders while providing recreational opportunities.

Community restoration plantings and carbon sequestration

Major land clearings from the past and present have made ecological restoration and afforestation an increasingly important activity. Remnants of forests, restoration plantings and naturally regenerating sites have the potential to not only connect the landscape and enrich its biodiversity, but to form part of the national carbon sink. Enrichment planting, weed and pest control, irrigation and proper management of these areas can result in higher rates of carbon sequestration (Forbes et al., 2020). Studying species composition and available carbon of existing community plantings with variable management will allow a better understanding of the carbon being sequestered (Ferretti & de Britez, 2006; Lu et al., 2018).

Native species, carbon content and biodiversity

All forests hold biodiversity, but some have higher amounts of it. Plantation forests have significantly lower quantities of native biodiversity when compared with native forests (Brockerhoff et al., 2008). When fragments of native vegetation are present in plantation forests' surroundings, the biodiversity levels rise (Deconchat et al., 2009). This increase highlights the importance of native remnants and plantings in terms of biodiversity levels. The higher levels of biodiversity provided by native forests support the need for these to persist within a landscape independently of the land use. Native forests also provide other ecosystem services such as carbon sequestration, soil protection, water quality, wildlife habitat, cultural values and recreational opportunities (MPI, 2018). All these benefits come together and highlight the importance of native forests within a landscape, but planting costs must be met as they are higher for native seedlings compared to radiata pine and other plantation forest species (Carver & Kerr, 2017).

Planting costs are a major limiting factor for native plantings. However, we still care for and plant native species, despite the costs, as interested landowners, local councils and community groups plant natives in preference over introduced species in many areas. Carbon credits are important to these initiatives as they can help offset these costs. The cultural identity and value of native species in areas of leisure and low productivity are far more significant. Biodiversity and humans have a worldwide connection; they provide us with resources, a sense of belonging, wellbeing, erosion control, and lower rates of water and air pollution, etc. (Chaplin-Kramer et al., 2019). In a landscape like New Zealand, where much of the indigenous landscape has been cleared and replaced, remnants can be connected and enhanced with proper management, including restoration (Norton et al., 2018; Norton & Reid, 2013). A per hectare of low biomass in planted or regenerated forest can be improved with proper enrichment planting of old-growth canopy dominant species, as there is a direct relationship between carbon biomass and biodiversity in lowland secondary successional forests (Carswell et al., 2012).

Enrichment planting

Carbon sequestration is only one of many ecosystem services provided by our forests. When planting non-forested areas, we immediately gain two distinct services, biodiversity enhancement and carbon sequestration (Carswell et al., 2012). These gains happen in local community plantings where a variety of ecosystem services such as the availability of taonga species, native habitats with high volumes of biomass, air quality, soil, and waterway protection from erosion in areas of cultural and recreational use are addressed. If we were ever to plant one species for carbon sequestration, we would be simplifying the complexity of our native forests by not establishing new ones and lose the overall benefits of desired ecosystem services for an extended initial period (Carswell et al., 2015; Graeme, 2001; Pawson et al., 2013).

The composition of community plantings vary due to seed source availability and species planted and often reflect the goals of the people behind the planting. Initial planting also often focuses on early succession species (e.g. genera such as *Coprosma*, *Pittosporum*, *Kunzea* etc) because they grow quickly and can establish a closed-canopy forest in a relatively short period of time. Because of this, few old-growth species (e.g. podocarps) are present and in the mature stage of these forests, the canopy cover of these species is often poor. In many restoration plantings, these are either absent or are present at a low number of successfully established individuals. Their importance relies on the successional stages of forests and when present they can contribute to higher biomass volumes (Williams & Norton, 2012). Understanding their growth, the total addition of carbon, and biomass volume will contribute to efficient planting for carbon sequestration. By interpreting the amounts of carbon present in different species, we will be able to identify differences between the contributions of short-lived species, such as *Coprosma robusta* and *Pittosporum tenuifolium*, to the overall carbon contents in mixed restoration plantings and contributions of long-lived species such as *Podocarpus totara* (Rüger et al., 2020). This interpretation will only show one piece of the puzzle, as the forests measured correspond to the early stages of the growth these forests present. The long-term carbon storage in these forests and species assemblages will not be represented by this study as mature forests will not be measured.

Measuring carbon content and allometric equations

Allometric equations are used to calculate the biomass of trees based on measurements, such as height and dbh from a live individual. These equations are developed using data collected from sample trees. These sample trees are measured in the field, then cut, weighed, oven-dried and weighed again. These steps give the species' wood density and a total dry mass of the tree which is then used to develop a mathematical relationship with the live tree measurements of height and diameter, allowing subsequent determination of tree mass without any further harvesting (Flores & Coomes, 2011). The mass, or oven-dry weight, is then transformed into carbon by dividing the result by 2 (Beets et al., 2012; Coomes et al., 2002; IPCC, 2003). This division is an international standard established by the Intergovernmental Panel on Climate Change (IPCC) of the United Nations.

The more trees included in the sample, the higher the accuracy of these equations in predicting live tree biomass. These equations can be national, regional or species-specific. They are used all around the globe with their biggest limitation being low sample sizes (Beets et al., 2012; Beets, Kimberley, Paul, et al., 2014). This is because cutting, drying and weighing trees is an extremely time-consuming process, especially for large trees. On top of this, finding trees to harvest is not always easy, and more so if limited individuals are available, such as in the case of natives and/or protected species. This is not a local problem; most countries have low resources, and available samples are a real challenge (Xing et al., 2019).

In New Zealand's restored areas, the native forest is replanted with a variety of species and under a range of site conditions, and the allometric equations used are not species-specific but designed for a mix of species. To allow for variation between species, a species coefficient or a mean species wood density is used because each species has different wood densities. Wood density for each species can also change depending on the region, water availability, altitude, ecosystem disturbances, species interactions and exotic plant invasions (Holdaway et al., 2017; Waller et al., 2020). However, it is hard to gather large samples to create or add information to allometric equations, and in some cases, fallen trees are even used. For this reason, allometric equations available for native plantings and native forests are limited.

The main allometric equation used for the national accounting of New Zealand's native trees is the *Natural Forest Allometric Equations for Live Trees* by Beets et al. (2012). Its sample size was approximately 140 trees, and it included the biomass of stems, foliage and large branches. This equation's sample comprised of 16 species, mainly of mature trees and some fallen individuals, with a single sample site in the South Island and six sample sites in the North Island. If using this equation for all trees and shrubs, the carbon estimation may be inaccurate as shrubs do not have the same biomass ratio between the foliage, stems and branches as a mature tree does. In 2014, a shrub equation was developed to address this (Beets, Kimberley, Paul, et al., 2014) based on height and diameter (at 10cm from the ground) measurements of live shrubs. This equation was based on two South Island sample sites where 21 species were included and 162 trees in shrub stages were harvested (Beets, Kimberley, Oliver, et al., 2014; Mason et al., 2014).

Thesis aim

This study aimed to investigate the live above-ground biomass (AGB) and carbon that is sequestered by planted indigenous forests on the southern Port Hills and Quail Island, Canterbury. Specifically, I will address the following questions:

1. How much CO_2 equivalent carbon is stored in restoration plantings of different ages and sites?

2. As potential future canopy dominants, how much CO_2 equivalent carbon is stored by planted tōtara?

To address these questions this research will focus on 15- to 59-year-old restoration plantings on the southern Port Hills and Quail Island.

This study will contribute to broader knowledge of growth and carbon sequestration rates of mixed restoration plantings, as we know little about secondary growth forests and how they develop in their early stages. Most available research around carbon sequestration in New Zealand is based on single-species native plantings, underrepresenting common restoration plantings. The MPI's native forest look-up tables used to quantify the amount of carbon sequestered do not represent the variability of mixed-species composition and their carbon uptake. Carbon credits may not reflect real carbon sequestration rates of these forests. This study will investigate species composition and its effect on carbon content in restoration plantings.

The national carbon accounting done by the Ministry for the Environment (MfE) uses a single carbon sequestration equation that is based on mixed mature forest individuals (Beets et al., 2012). This study will compare the use of this equation and the use of a specific shrub equation (Beets, Kimberley, Paul, et al., 2014). This study assesses how the carbon calculated by the MfE is representing young secondary forests in their reports to the UN.

We also do not know if any species combination sequesters more carbon than others. This study assesses tōtara's carbon content, to see if the most frequently canopy dominant species present in the study areas can contribute to a significant amount of carbon within these plantings. This assessment will inform us whether adding long-lived canopy dominant species to enrich these areas will benefit their biodiversity and their carbon uptake.

This study will contribute to the understanding of how much CO_2 equivalent carbon can be expected to be stored in planting native trees to restore the southern Port Hills and Quail Island. The results of this study will help us understand the current carbon status behind the work volunteers have invested in the area, and what carbon amounts are held in these forest types. Results will contribute directly to local stakeholders involved in the process of restoration and carbon sequestration, such as the Te Kākahu Kahukura, a collaborative group that drives efforts towards plantings and restoring sites of native vegetation, connecting and enhancing biodiversity around the southern Port Hills, Quail Island and beyond. Landowners and projects linked to Te Kākahu Kahukura will be able to see the variability of existing restoration efforts and join them in a comprehensive vision that will produce a carbon sequestration impact on the southern Port Hills and Quail Island. This effort can also contribute to plantings from other important stakeholders, such as the Christchurch City Council (CCC) and their 2030 carbon-neutral plan.

CHAPTER TWO – Study Area and Methodology

Study area

Port Hills

The Port Hills is a unique environment that once was covered in dense podocarp-broadleaved forest, but almost all this forest was lost as a result of Polynesian and European deforestation. Since the European settlement and changing land uses, the current vegetation cover of the Port Hills has developed, which is a mixture of pasture, plantation forests, regenerating native forest, native and exotic shrubland, restoration plantings and tussock grasslands. This area has a variety of uses including sheep and beef farming, plantation forestry, recreation, lifestyle blocks and forest restoration (Carter, 2003). The land is in a mixture of tenures, including public and private.

The cultural value of this site for Christchurch and the surrounding communities is of high importance as it is a hub for the city's recreational habits and contains some of the best remaining natural vegetation. It has a natural heritage unique to its geological features, as it rises from sea level to 573m at its highest point, with a combination of mainly basaltic and trachyte rocks from remnants of the Lyttleton volcano dating back 10 to 12 million years (Wilson, 1992).

Small remnants of native forests comprise what once were dominant species in the Port Hills landscape such as *Podocarpus totara* (tōtara), *Prumnopitys taxifolia* (matai), *Dacrycarpus dacrydioides* (kahikatea), *Melicytus ramiflorus* (mahoe), *Griselinia littoralis* (broadleaf), *Pennantia corymbosa* (kaikomako), *Hoheria angustifolia* (narrow-leaved ribbonwood), and *Plagianthus regius* (lowland ribbonwood) (Wilson, 1992). Most of the planted and regenerating areas are comprise of *Kunzea robusta* (kānuka), mahoe, *Pittosporum* species, and *Coprosma* species, as well as a range of exotic woody species (especially *Ulex europaeus* and *Cytisus scoparius*). These restoration efforts date back to 1903, when Harry Ell became a Christchurch City Councillor. His work around the Port Hills involved creating the summit road, the Port Hills Rest Houses and four scenic reserves (Oakley, 1960). From his legacy, the Summit Road Society was born in 1948, a community group that provides ongoing restoration in the area (Ogilvie, 2009). His initial restoration work (Kennedys Bush) is also under guardianship of the CCC, which has continued working in reforesting the Port Hills with native trees.

Quail Island

Otamahua (Quail Island) is a recreational reserve south of Christchurch comprising 85 ha under the administration of the Department of Conservation (DOC). Since 1998, the Otamahua/Quail Island Restoration Trust took the role of eco-restoring the island according to an original plan created in collaboration with DOC, Ngati Wheke, ecologists from Lincoln University and Canterbury University, Landcare Research and other professionals in the area. This collaboration arose from Ray Genet's Master research thesis (Genet, 1997). The shared interest behind restoring this island is based on historical records of its high biodiversity land cover, comprising a rich indigenous forest ecosystem that has since been disturbed with deliberate clearing by fires and introductions of new woody and productive agricultural species (Burrows et al., 2011). A restoration plan was created with long- and short-term goals that have since been carried out, and has led to a current successful secondary lowland regenerating forest (Genet & Burrows, 1998).With only two predators left, mice and the occasional deer that manage to swim across from the mainland (Bowie et al., 2018; Genet, 1997), this area has the uniqueness of trees growing with less pressure from pests and grazing animals than is the case on the Port Hills.

The trust has carried out a continuous effort of care, plantings and maintaining records of their work, which make this site an ideal place to study the results of restoration plantings. With the land's future to be used as a permanent forest for recreational use, it is an ideal area for this study to measure the carbon content of mixed native plantings.

Study sites

Tai-Tapu

The Tai-Tapu site is private land located at the base of the Port Hills near the Tai-Tapu village. The site underwent restoration planting by the previous owner as part of a resource consent application in 2001 for a subdivision proposal (Table 1). Currently, much of the forest has been regenerating around the planted area with native species (e.g. Kānuka), gorse and broom. Two main areas have well-established indigenous plantings and were sampled in this study (Figure 1).

Quail Island

The oldest planted sites on Quail Island were used in this study due to time constraints and the size of the trees. These were mixed species that were planted by volunteers between 1999 and 2000 (Table 1, Figure 1). The access to the sites is easy, and maintenance is still being carried out by the Otamahua Ecological Restoration Trust staff and volunteers.

Kennedys Bush Reserve

The areas that were measured were identified as plantings undertaken by the Christchurch City Council and local community groups, such as the Summit Road Society. The dates of these plantings ran from 1951 to 2001, with information on their age retrieved by the Christchurch City Council staff and in collaboration with the Forestry Science Master's thesis of Stephen Reay, who later confirmed the sites through aerial images (Table 1 and Figure 1).

Ohinetahi Reserve

The Summit Road Society manages all plantings in this reserve. The sites which are the oldest plantings on the reserve ranged from 1999 to 2005 (Table 1). The site is currently still under the same administration. No further plantings are foreseen in the future, although continuous management of existing plantings will occur (Figure 1).

Hoon Hay Reserve

These are three, small planted areas located along the top of the reserve, all facing north and away from the natural regenerating areas. These plantings were undertaken thirty years ago by Dr Colin Burrows and are currently under the care of the CCC (Table 1 and Figure 1).

Table 1: Study sites characteristics.

Site	Number of plots	Elevation range (m.a.s.l.)	Planting age range	Grazing animals	Slope range (°)
Tai-Tapu	2	414-459	19	Pigs, deer, sheep, rabbits, mice, stoats, possums	15-21
Quail Island	9	34-129	20-21	Deer and mice	4-22
Kennedys Bush Reserve	7	450-492	20-59	Pigs, deer, sheep, rabbits, mice, stoats, possums	20-35
Ohinetahi Reserve	3	161-221	15-21	Mice and possums	13-24
Hoon Hay Reserve	4	414-459	30	Deer, sheep, rabbits, mice, stoats, possums	2.5-24

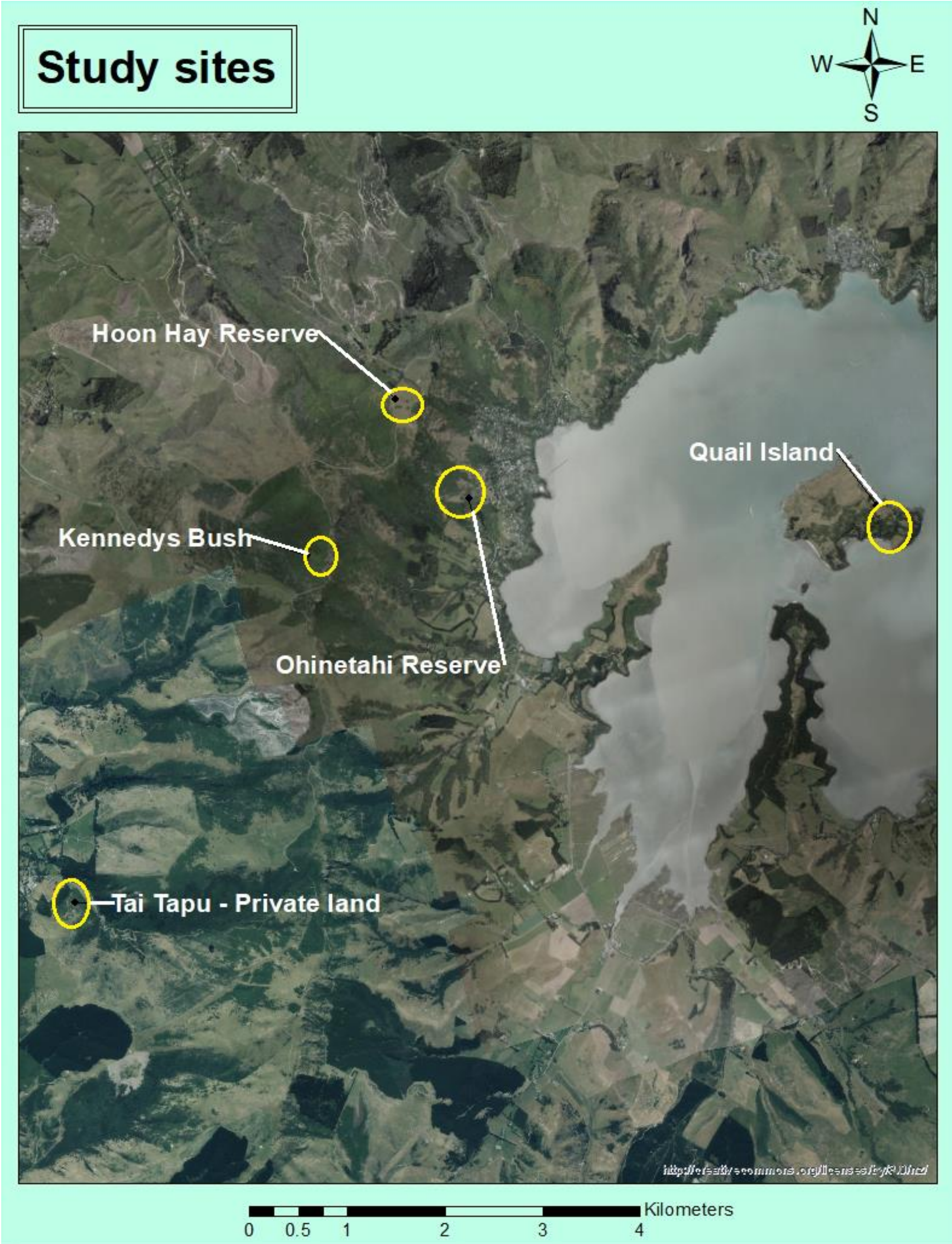

Figure 1: All areas in yellow where plots were placed in the 5 study sites.

Methodology

Carbon content in biomass on the southern Port Hills and Quail Island was measured. Twenty plots were established as sampling units in restoration plantings where approximate estimates of planting age were available. In addition, the native planting areas studied had to consist of 70% native plantings and 60% woody cover to avoid excessive patchy areas and allow a clear representation of a restored forest. No ethics approval was required to undertake this study; however, an agreement with landowners, restoration managers and stakeholders was necessary for site access.

The randomisation of plots

The sample plots were located through a simple layout of a grid of points, evenly spaced and randomly located over a satellite image that covered each of the study sites. This step was done using a shapefile of all the boundaries of the native planting areas, this was created over a satellite image from the Land Information New Zealand Data Service (online connection of the Canterbury maps). Each point was a location on a 20x20m grid. The spacing was done to increase the chance of plot locations occurring in smaller plantings. The origin of the 20x20m grid was randomised, and plot locations in larger sites (Quail Island and Kennedys Bush) were alternatively measured to allow enough space between plots. The alternation made it closer to a 40x40m grid. The selection of alternant points was made randomly by appointing the initial plot point before arriving at the site and once measured in the field the subsequent point was measured. This alternation was necessary due to time constraints and similarities between plots (Figure 2).

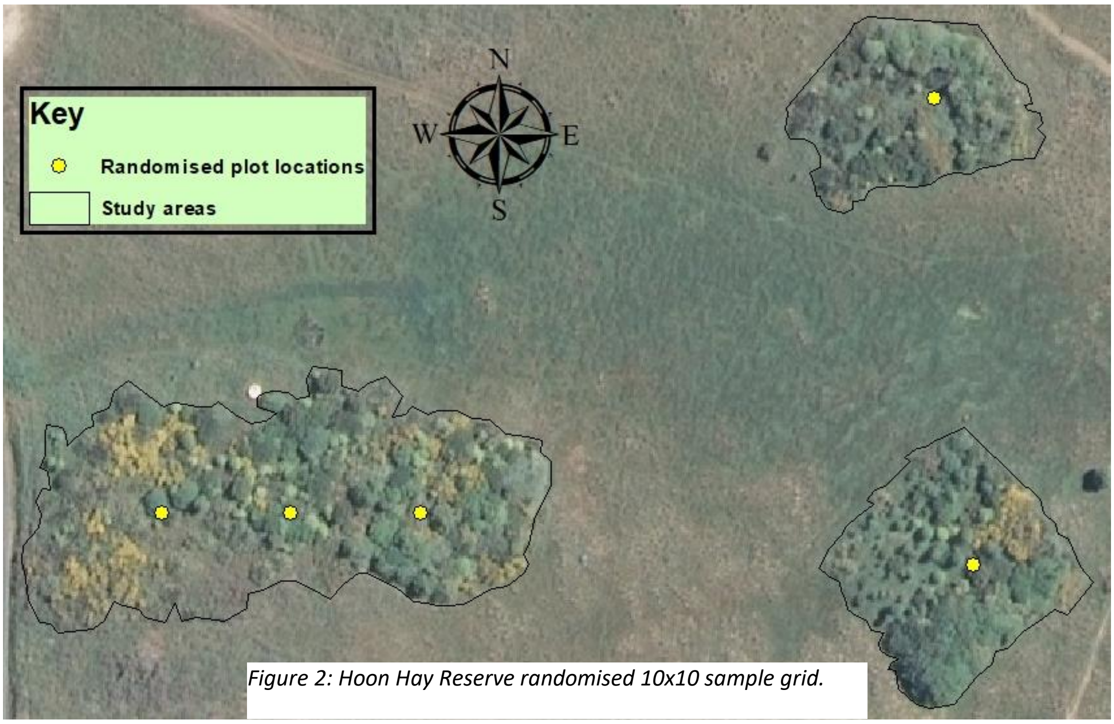

Figure 2: Hoon Hay Reserve randomised 10x10 sample grid.

The location of each plot was determined in the field using a Garmin GPS with New Zealand Transverse Mercator (NZTM) system, which was used to locate the centre of the plot. Due to satellite imprecision under a dense canopy, a constant error of up to 4 meters was observed when locating a plot's centre. This error was tested on Ilam Fields, an open grass area with good satellite reception. To minimise and avoid close sampling of sites, a waypoint was marked on-site where the estimated sampling point was obtained in the field. Plots were always offset by at least 2 meters from the edge and a 10 meter distance from other plots, to maintain a constant separation in the sample sites. This method was based on the stratified random sampling technique of the Intergovernmental Panel on Climate Change (IPCC), where its main objective was to lower variability within the samples as well as reduce the possibility of bias and incrementing the objectivity of the study's sampling (Bickel et al., 2006).

Plots

The plot layout was based on an adaptation of the Department of Conservation Tier 1 field protocols with the biggest difference being the use of temporary sample units instead of permanent plots, and a reduction of plot size to 10x10m instead of 20x20m. The smaller plot size was considered appropriate given the generally small size of planted species at the sites. At the same time, their temporality was permission-based as well as aligned to the research objective of measuring current carbon content. A similar method has been used by the Farming and Nature Conservation Project where plot size was also 10x10m, allowing them to better represent remnant areas of native forests (J. Foster, personal communication, January 10th, 2020).

Plots were subdivided into four subplots of 5x5m to facilitate the measurement and identification process. This methodology improves the accuracy of the plot dimensions by setting tapes tightly on the ground with pegs. These tapes were also used to measure the length and width of the plots. A Vertex was used with a Transponder on a Monopod to check that a diagonal of 7.7m was met in every 5x5m subplot. This methodology, in combination with a 3-4-5 method, was used to confirm the angles of the plots were 90° (Figure 3). This technique consisted of measuring 3cm along the exterior tape starting from a corner peg and on the adjacent tape measuring 4cm. The distance between these measurements was 5cm on a 90° (Figure 4). This ensured that plots were very accurate which is important for gaining an accurate estimation of CO_2 equivalent carbon present.

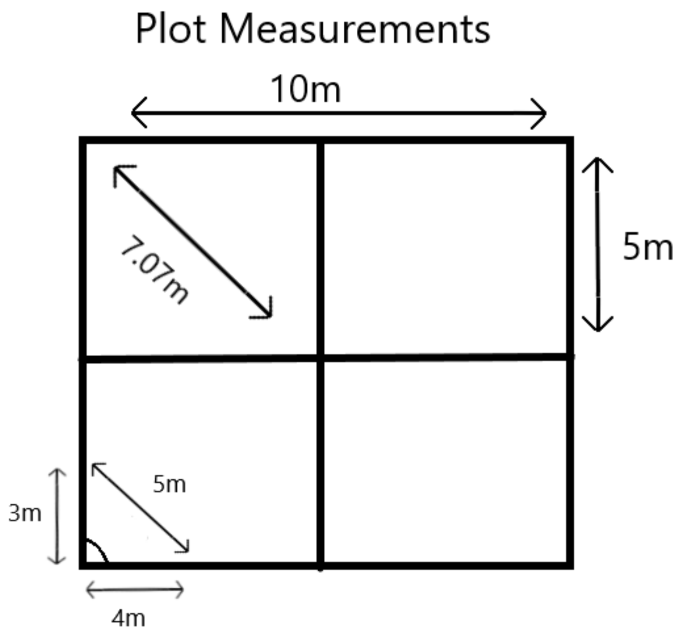

Figure 3: Plot measurements for Total plot, Subplot, subplot diagonal and the 3-4-5 method.

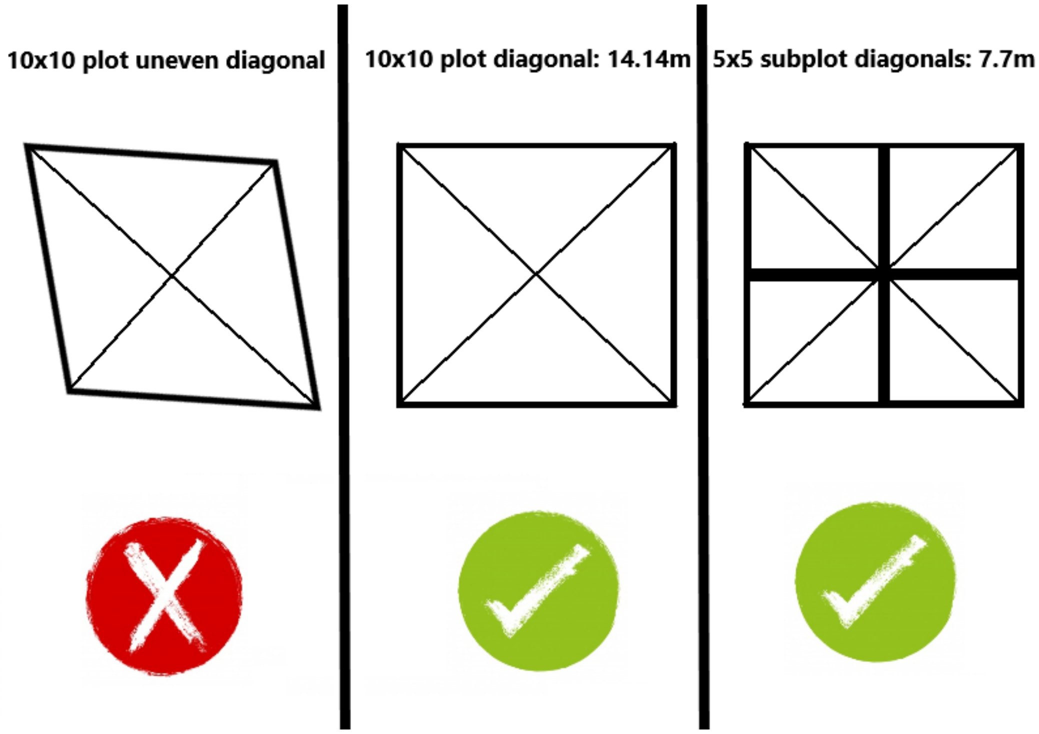

Figure 4: The importance of 90° angles

Field instruments

All instruments used on the field except for the pocketknife were provided by the University of Canterbury (Table 2). The recording sheets had been adapted specifically for this study and were based on the DOC Tier 1 protocols.

Table 2: Field instruments

2 x 80m tape	70cm metal stakes
2x 60cm diameter tape	Camping stakes
Sighting compass	Garmin GPS (Montana 650t)
Measuring pole with tape	Fluorescent tape
Clipboard and pens	Recording sheets
Extra batteries	Plant ID Books
Clinometer	Vertex + Transponder + Monopod

Field measurements and allometric equations

In the field, the slope was measured, and plots were laid as 10x10m across the slope. When calculating the carbon content for each plot, the totals were corrected. This correction is for all results to be proportional to their horizontal area. To accomplish the correction, differences with their horizontal distance were added and calculated using the proportion of the missing area corresponding to the slope measurement. This was done and added as the equivalent carbon measurement of the plot.

Live ABG measurements for trees, shrubs and saplings

A single measurement was taken for the diameter at 10cm above the base of the stem, and it was recorded as D. If the tree or shrub was forked, then the stems were considered as separate individuals and recorded as such. This was because at less than 10cm from the stem base it becomes challenging to determine if an individual is part of the same plant or a different one (Beets, Kimberley, Paul, et al., 2014).

A second diameter measurement was taken to account for the individual's diameter at breast height (dbh) if it was forked above 10 cm, then all stems ≥ 2.5cm were measured. This measurement was taken at 1.35m from the base of the tree as per the international standard and Tier 1 protocol to measure carbon sequestration (DOC, 2017). Dead stems still attached to the tree were not measured, as per the objectives of this study.

A single height measurement was taken from the tallest stem of the tree. Where possible, the Vertex was used for this measurement however, due to the occurrence of dense canopies, a measuring pole was the main tool to use in these cases. When trees were growing at an angle, their height was taken from ground level to its crown following the stem's growth direction.

All live trees ≥2.5 cm dbh over bark were measured; anything lower was recorded as a sapling. Saplings were only counted and not measured. They are important for this study as they are the biggest indicators we have of these forests' regeneration potential.

Trees, shrubs and sapling recordings

Individuals were considered as trees when their dbh was ≥10 cm, and they had a height of ≥ 1.35 cm from the ground. Shrubs were defined as having between ≥ 2.5cm and < 10 cm dbh and height ≥ 30cm from the ground. Anything < 2.5 cm dbh and < 30 cm in height was recorded as a sapling.

Because most of the individuals' stems were forked and multiple dbh measurements were taken, the classification of tree or shrub for each individual was known when the sum of the volumes of each measurement was calculated and added off-site for the dbh (Equation 1).

$$dbh = \sqrt{\sum_i dbh_i^{\,2}} \qquad (1)$$

Distinctions between shrubs and trees are essential as they were applied for field measurement and the biomasses were calculated using two different allometric equations. The shrub equation used G, and the tree equation used the dbh measurement.

Quadratic measurements

Discrete plants are individuals whose measurements were difficult to obtain and cover a considerable area within the plot; they are species that don't classify as trees, shrubs or saplings. The case of native flax or vine species was recurrent. The protocol followed by the Tier 1 in these cases was used. Two orthogonal distances were taken of the individual (perpendicular measurements as seen in Figure 5), as well as a height measurement (DOC, 2017). A quadratic allometric equation was used to estimate the measurements of these individuals (Holdaway et al., 2017).

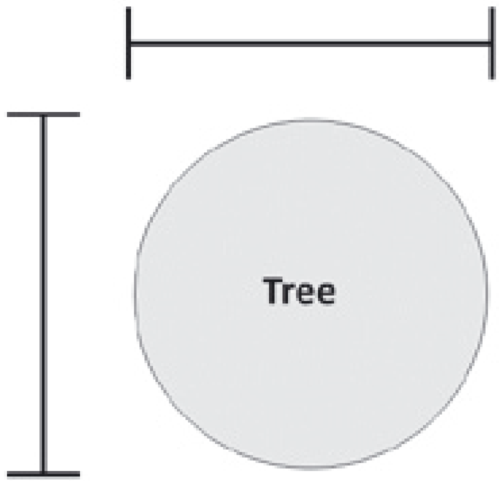

Figure 5: Orthogonal measurements of an individual, using a 2 m tape (DOC, 2017).

Species identification

Identification of species in plots was made using native identification books and the New Zealand Conservation Network website as well as the app iNaturalist. This information was later used to understand the species assemblage of the areas of study. The current National Vegetation Survey (NVS) names were used to identify and record the species type on-site and transferred into an Excel file with all the data gathered.

Data and analysis of results

Firstly, all data collected were used to quantify the amounts of carbon sequestered using two different allometric equations for mixed native species. One of the equations has been based on mature trees and the other one for shrubs. Differences between the use of each of these equations (mature tree equation and shrub equation) was looked at for all individuals measured. The results showed a variation of the total uptake as a reflection of the equations used. Secondly, the carbon content amounts were combined to give an overall carbon content per hectare. This showed the variation in the total amount of carbon sequestered in different mixed-species plantings since time of planting (along with the small initial carbon in planted seedlings), as these were characterised and grouped by various factors such as age, site, elevation and aspect. Thirdly, calculations were done for the total carbon sequestered per tōtara tree growing in different surroundings such as single standing, around gorse, within restoration plantings or by the restoration plantings edge. With these values, a representation of the variation within the total amounts sequestered was used to identify and understand if the characteristics of the surroundings may vary the carbon content of this species.

In the sense of a plantation forest yield model, no yield model was developed in this study, as I aimed to understand variation in carbon content within the restoration plantings' composition and tōtara trees. I looked at carbon present and identified what factors such as plot location, species diversity and time from planting had a significant effect using statistical modelling. A tree model was developed, and sigmoidal curve models were applied to the data where Gompertz and Schumacher equations were fitted (Hong et al., 2004; Luo et al., 2018; Payton et al., 2009).

<u>Assumptions</u>

Tree and shrub size were assumed to follow each allometric equation. As indicated by each equation's authors, tree and shrub growth stages for all species were assumed to be represented by the standard sizes recorded in Table 2.

The allometric equations implied that their coefficients were representative of the species found in all twenty- five plots and of single measured tōtara. The application of wood density values from the Landcare research database is assumed to represent all species. The values of wood density used for individual tōtara are also considered to represent the 105 individuals measured.

The year of planting for all species within a plot was assumed to be the same. Possible natural regeneration within the plots was not investigated as they are considered part of their natural succession. The selection of proximity to evident natural regeneration areas was conservative when mapping for sampling sites to avoid any date conflict.

<u>Summaries of the Result Chapters</u>

Chapter 3.

A bibliographic overview of the different allometric equations used in New Zealand for native species CO_2 equivalent calculations is undertaken. I also assessed whether the use of a mature forest allometric equation (Beets et al., 2012) is enough to calculate all shrub and trees species AGB, or if an additional equation specific for native shrubs (Beets, Kimberley, Paul, et al., 2014) is better practice in these young regenerating forests.

Chapter 4.

An analysis of the carbon amounts of the areas measured and their relationships was undertaken. The data on species composition and possible environmental effects over the amounts of carbon sequestered was also looked at.

Chapter 5.

A representation of the carbon amounts that individual tōtara trees currently have on the southern Port Hills and Quail Island was undertaken. I also looked into the differences and contribution of tōtara carbon content incurred in these plantings.

Chapter 6.

Discussion of all the results, a conclusion and implication of this study are addressed.

CHAPTER THREE – Results: Allometric Equations

Introduction

As restoration plantings, especially when young, have a large proportion of plants that can be described as shrubs, it was deemed necessary to compare across the sites sampled here the effects of using an allometric equation based on mature senescent trees (used by the MfE national reporting system to the UN) and an allometric equation based on shrub-stage trees. This chapter provides this comparison and is used to formalise the equation used for the rest of the research.

Methodology

All diameters and heights of live trees were measured in 25 random plots on Quail Island and the southern Port Hills (see last chapter for details). This included trees, shrubs and discrete plants; species that grow in clumps such as harakeke or vines that do not have a tree-like or shrub-like growth form. The data collected was used to calculate the volume of each individual. The volume was then converted into carbon by multiplying its value by the species-specific wood density and dividing the outcome by two. This resulted in an individual estimate of carbon for each species. To account for total above ground carbon (AGC), all carbon values were summed up at a plot level (discrete plant, shrub and tree carbon). A slope correction was then used to account for slope. In addition to this, it is common practice for a below-ground carbon amount of 25% to be added as per the IPCC standardisation. Because no significant studies have been able to clarify accurate amounts of underground carbon content of New Zealand native forests due to the labour-intensive work associated with digging roots, below-ground carbon will not be accounted for in this research. The focus of this study is to clarify what carbon was present in live above-ground biomass.

Data preparation and error checking

All field measurements were entered into a data file and verified to satisfy the sizes of height and stem diameters for classifying trees and shrubs (Table 3). These classifications are based on both allometric equation parameters. For creeping species, such as *Muehlenbeckia complexa*, or species where no diameter was able to be measured, such as harakeke (*Phormium tenax*), a cuboid equation was used, and size limitation was based on height. These species are classified as discrete plants and usually grow as clumps.

Table 3: Minimum measurement for trees and shrubs to calculate carbon content using native forest mixed-species allometric equations (Beets et al., 2012; Beets, Kimberley, Paul, et al., 2014)

	Height (m)	DBH (cm - at 1.35cm from ground)	D (cm - 10cm from ground)
Discrete plant	≥1.35	-	-
Tree	≥1.35	≥ 10	
Shrub	≥ 30	10 > DBH ≥ 2.5	≥ 2.5

For individuals that had multiple stems, the total volume was calculated using Equation 2. This allowed calculation of one dbh value for each individual.

$$dbh = \sqrt{\sum_i dbh_i^{\ 2}} \qquad (2)$$

<u>Biomass calculations</u>

Tree equation:

To calculate the natural forest AGC in biomass (Equation 4), the species wood density (SWD) was multiplied by the ratio estimation of each species and assigned to each tree (D_{stem}). The use of a ratio value is to improve the estimates of the whole stem plus branch wood density differences by species and was obtained from Beets et al. (2012). Species-specific wood density values were taken from available New Zealand literature (Clifton, 1990) and the Landcare Research database, containing values for 114 different species. Where values were not available, it was determined based on the average of their genus. When no other data for the genus was available, wood density was based on the average of all New Zealand wood densities (=502.2708) (Coomes et al., 2014; Flores & Coomes, 2011; Holdaway et al., 2017).

The volume of the stem and branches (V_{stem}) is calculated from Equation 3. With the volume, the ABC is then calculated using Equation 4.

$$V_{stem} = 4.83 \times 10^{-5} \times (DBH^2 \times H)^{0.978} \qquad (3)$$

$$C = 0.5 \times D_{stem} \times V_{stem} + 1.71 \times 10^{-2} \times DBH^{1.75} \qquad (4)$$

Shrub equation:

The D values were transformed into square meters (Equation 5) before calculating carbon content (Equation 6). Additionally, each individual was appointed a species-specific coefficient ($a_{species}$) and for those where values were not available, an average species coefficient was used (184). This coefficient replaced the use of a species wood density value. The result (*dry weight*) from Equation 6 was divided by 2 to get the total carbon content.

$$G(m^2) = (\pi \times D \div 100 \div 2)^2 \qquad (5)$$

D = Diameter at 10 cm from the ground (cm)

$$Dry\ weight\ = a_{species}\,(G\ \times H)^{0.837} \qquad (6)$$

Discrete plant equation:

Carbon amounts of discrete (individual or clump), where orthogonal widths and height were taken, were calculated based on cuboid geometry (V_{shrub}) and density based on published values from destructive sampling (Carswell et al., 2014; Coomes et al., 2002). This relationship (Equation 7) was later converted into carbon by dividing its value by 2.

$$Dry\ mass = density \times V_{shrub} \qquad (7)$$

Slope and plot carbon

The carbon of all individuals was summed separately by the plot for the values of trees, shrubs and discrete plants. The slope was then corrected for every plot by calculating the cosine of the slope angle and adjusting the carbon difference that each plot had to its horizontal plane. This means that the percentage missing of carbon by its slope difference to its horizontal plane was added.

Results

The total carbon calculated for all individuals measured, showed a difference when using the shrub equation in comparison to the use of the mature tree equation. The results showed an overall higher estimation when using the tree equation (Figure 6). This difference was minor and could reflect the tree equation accounting for the carbon of large branches and foliage versus the use of a species coefficient for the shrub equation that accounts for an overall tree form. Regardless, the carbon accounted for all species using both shrub and tree equations as variables present a linear relationship (p-value = < 2.2e-16) (Figure 6). This gives us an indication that the carbon calculated using either equation is relatively comparable, except for plot 7 (Table 4). This plot was measured in Kennedys Bush, in the second oldest planted site, comprising three mature *Olearia paniculata* individuals with significantly smaller dbh values in relation to their D measurements. This reflects an occurrence in the site where large stems of multi-stem trees were decaying. This could be explained by the presence of deer in the area who stress the bark of trees by rubbing their antlers, resulting in decay. Bare sections of bark were observed in more than one *Olearia paniculata* within the site's proximity.

In the case of discrete plants, these were only present in 12 plots where harakeke was the main species. The high values in plots 3 and 10 (Table 4) correspond to the dominance of harakeke within these plots. As harakeke is a common species in restoration planting, it is important to acknowledge its presence and account for its carbon.

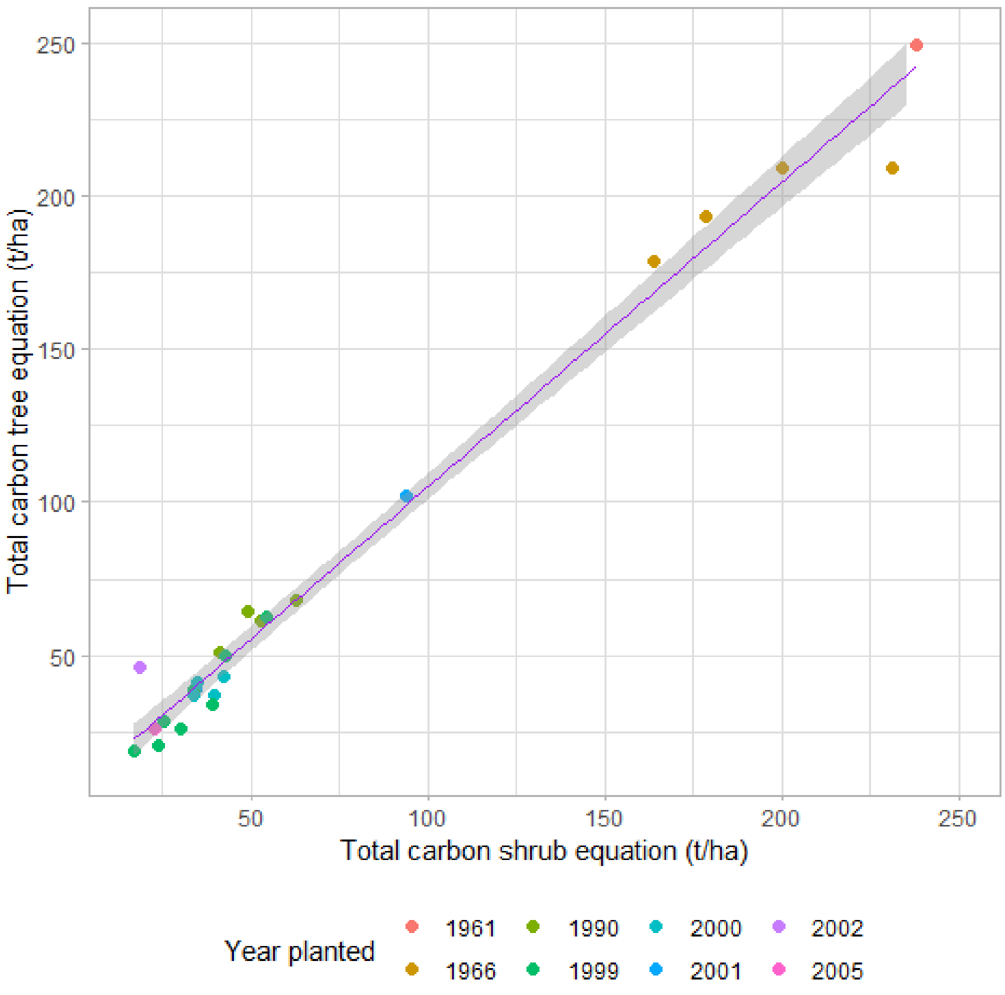

Figure 6: Total carbon calculated using the shrub equation and tree equation by year of planting.

Table 4: Total carbon for discrete plants, all individuals based on the shrub equation (Beets et al., 2014) and mature tree equation (Beets et al., 2012) in t/ha for each site, plot and year of planting.

Plot	Site	Year of planting	Discrete plants	Shrub Carbon	Tree Carbon
1	Hoon Hay Reserve	1990	4.05	36.86	46.77
2	Tai Tapu-Maury	2001	0.06	34.12	38.43
3	Hoon Hay Reserve	1990	6.34	56.45	61.72
4	Quail Island	2000	1.08	38.22	35.99
5	Kennedys Bush	1961	1.84	236.15	247.48
6	Kennedys Bush	1966	0.46	199.76	208.55
7	Kennedys Bush	1966	0.1	230.85	209.4
8	Kennedys Bush	1966	1.53	162.21	177.49
9	Kennedys Bush	1966	0.04	178.68	193.38
10	Quail Island	1999	6.06	19.5	22.25
11	Quail Island	1999	1.12	15.97	17.81
12	Hoon Hay Reserve	1990	1.27	51.36	60.34
13	Quail Island	1999	NA	33.64	38.9
14	Hoon Hay Reserve	1990	NA	49.02	64.52
15	Kennedys Bush	2000	NA	33.65	37.18
16	Kennedys Bush	2000	NA	34.69	40.98
17	Ohinetahi Reserve	2005	NA	22.68	25.99
18	Tai Tapu-Maury	2001	NA	93.91	102.36
19	Quail Island	1999	NA	23.98	20.4
20	Quail Island	2000	NA	42.38	42.97
21	Ohinetahi Reserve	1999	NA	54.48	62.39
22	Quail Island	1999	NA	39.11	34.15
23	Quail Island	1999	NA	30.02	26.09
24	Quail Island	1999	NA	42.47	50.02
25	Ohinetahi Reserve	2002	NA	18.45	46.33

To assess differences in the use of a shrub equation versus a tree equation when quantifying the carbon content of secondary forests, the data was graphed as a histogram with a scaled power transformation to better interpret the results (Figure 7 and 8), (Karian & Dudewicz, 1999). The distribution of both histograms is negatively skewed (less homogenic leaving a longer tail of lower values), and the overall values are predominantly higher than the mean, this is more significant for the tree equation (Figure 8). This may well reflect the age difference between all the plots and the carbon that is sequestered over time. Meaning the data has a low number of samples for a high number of sampled years.

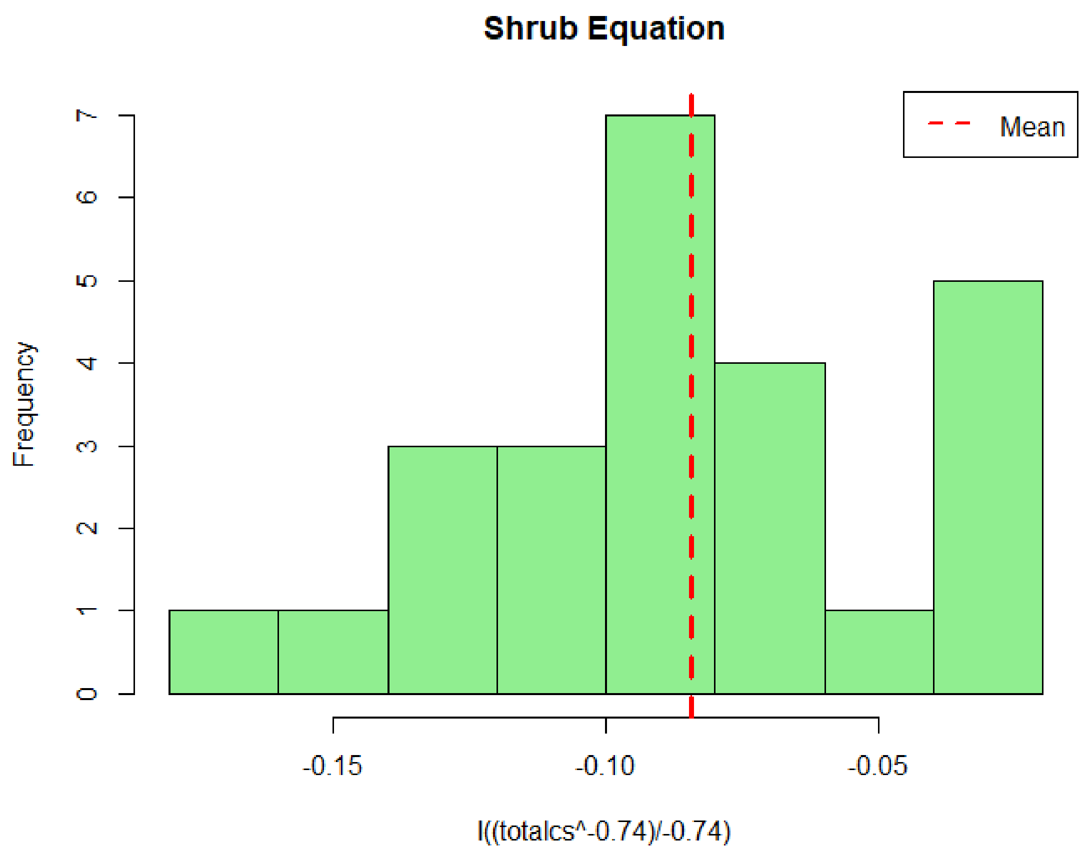

Figure 7: Histogram of the total carbon using the shrub equation.

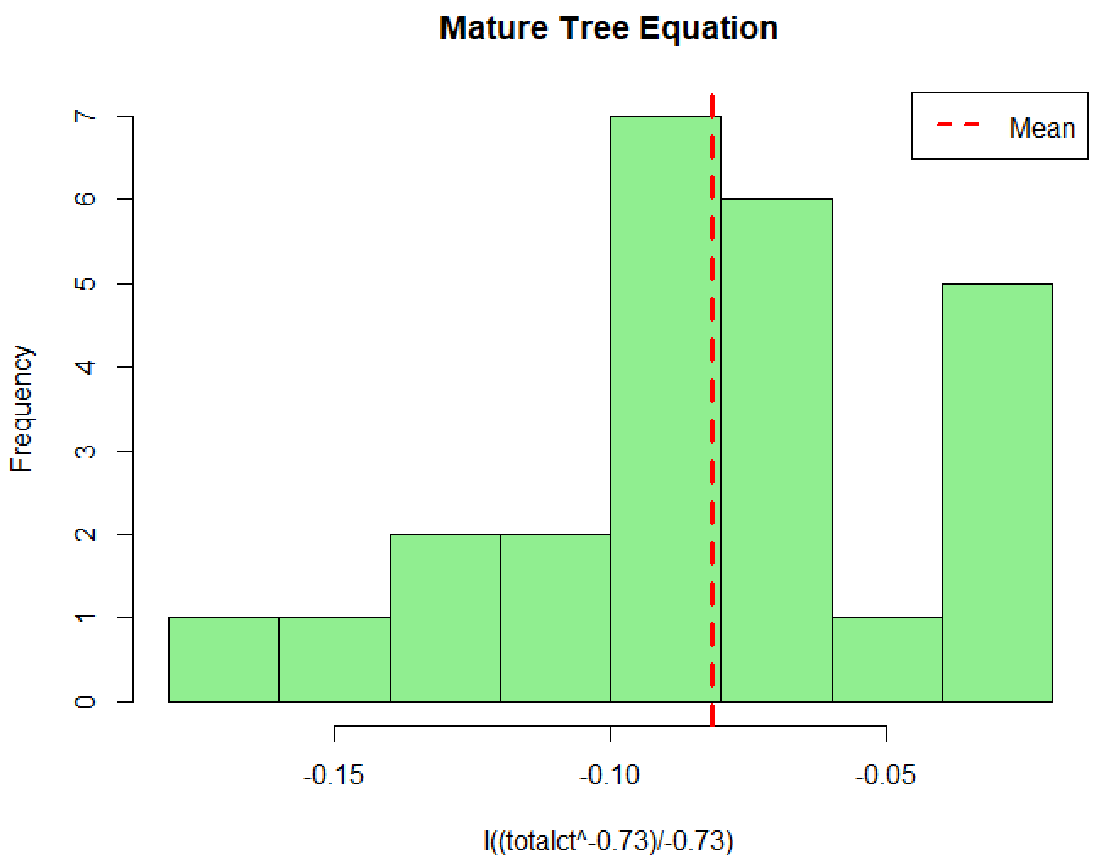

Figure 8: Histogram of the total carbon using the tree equation.

While over the 60 years covered by the samples, sequestered carbon increased at a relatively even rate, there is considerable variability in the first 20 years (Figure 10 and 11). This is in part due to an outlier, plot 18; a 2001 restoration site located on a highly productive site.

During the first 30 years of planting, the carbon content shows low changes in their values the older the plantings are; after that, a notable difference is observed (Figure 9 and 10). This is seen independently of the equation used. However, a larger difference in the use of a mature tree equation compared to the shrub equation is seen for plots that are 54 years and older. This emphasises the importance of using the mature tree equations for older plantings.

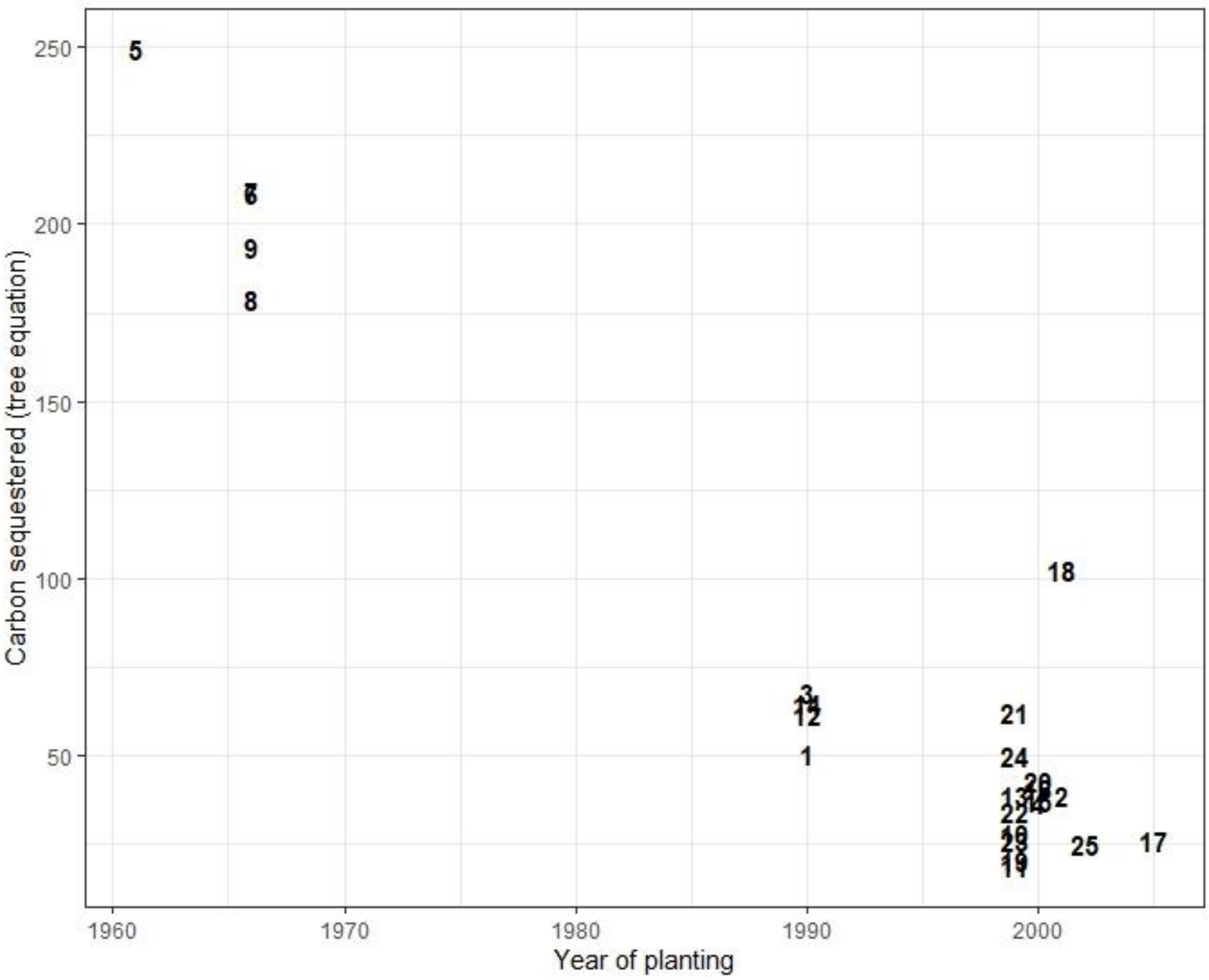

Figure 9: Carbon sequestered using the tree equation by year of planting in all 25 plots. Numbers represent different plots.

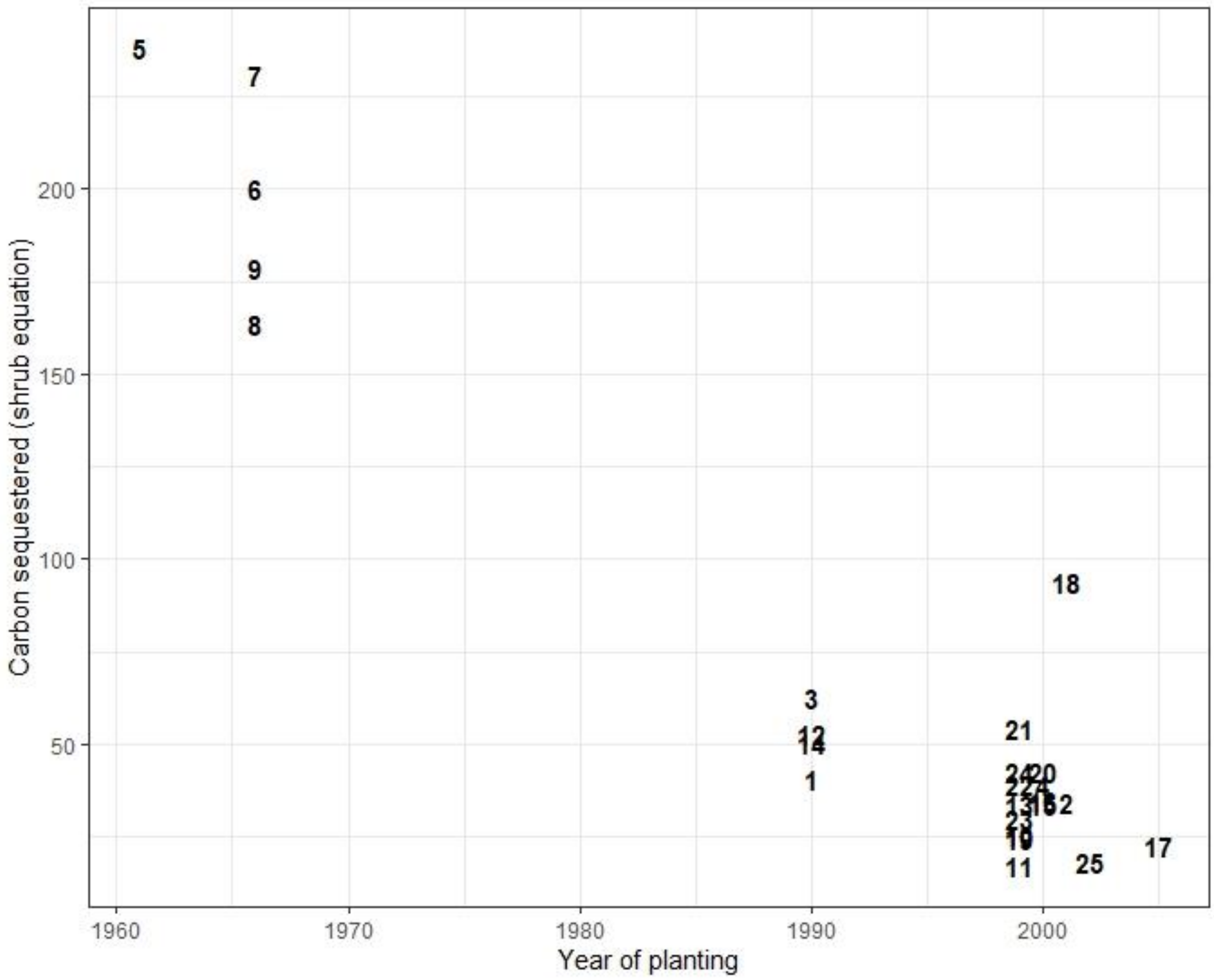

Figure 10: Carbon sequestered using the shrub equation by year of planting in all 25 plots. Numbers represent different plots.

Discussion

The amount of carbon present in plots were similar between plots planted in similar years. Exceptions to this were the two plots planted in 2001 (Figure 11), where the amount of carbon was close to double for plot 18 than plot 2 (Figure 9 and 10). The possible factors to be considered for this are their environmental aspects; the altitude, physiography and soil of both differ substantially. Plot 2 is located up the slope at 133m elevation, has dry soil and a concave shape. The bottom plot (18) was at 43m elevation and was located at the bottom of the slope in a valley where shade persists for longer and water may accumulate after heavy rain. As plot 18 exhibits an unusually high amount of carbon, it reflects how high productivity land usually excluded from native restoration provides favourable conditions for tree growth and hence carbon sequestration.

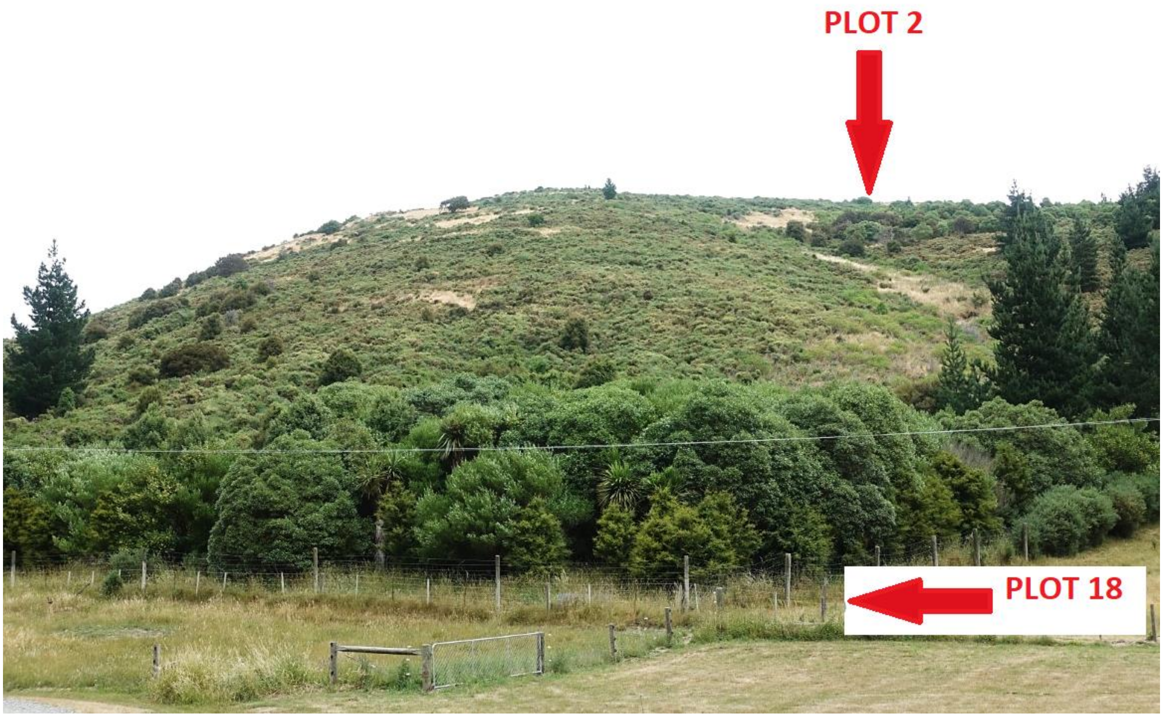

Figure 11: Tai-Tapu 2001 restoration planting showing the site where plot 18 is located close to fence and bottom forest, and plot 2 at mid hillside where a few trees appear amongst gorse in the upper right of the picture, (unpublished picture by David Norton).

Another interesting result is seen in the carbon of plot 3 (Table 4). Its values were unusually high for the low number of trees and shrubs present, mainly due to a high amount of carbon for discrete plants due to the dominance of harakeke within the plot. This can be explained by the coverage of harakeke within two of its subplots where no trees grew, hence allowing light to reach the understory and forest floor in the middle of the planting (Figure 12). The presence of gorse and grass was observed as well as higher diameters for trees growing next to the harakeke clump. A possible edge effect could explain the high-volume biomass of the trees due to the light reaching the middle of the planting area (Ranney et al., 1981; Reinmann & Hutyra, 2017). The same effect could be observed in plot 10. Still, the biomass of the trees and shrub equations did not show high biomass, and this may be because the canopy was still very young. Trees were growing between the harakeke therefore not allowing enough sunlight to reach the understory. All these factors may have contributed to carbon content of the same year plantings resulting in marked differences during the first few years of growth (Reinmann & Hutyra, 2017). However, once the trees over-top and shade the harakeke as the plantings get older, this effect will disappear.

Figure 12: Plot 3 edge and opening of the centre of the planted area. Harakeke clump growing next to gorse, grass and trees.

Overall, both allometric equations showed an increase in carbon present with planting age. These increases were higher when using the tree allometric equation. The increases were not constant for all plots and were likely to be related to the different species present, the stem sizes and /or environmental conditions. On average the carbon content results were 9% higher when using the mature tree equation than using the shrub equation. The use of an equation based on mature trees for all individuals planted may be appropriate if the cost and time involved in taking another measurement of the tree (G) and processing data is not significant to a specific project. In the case of the national scale having a possible percentage difference of 9% within the accounting for all New Zealand's secondary growth forest, it is well worth the effort of ground proofing these equations. The ground proofing would involve sampling harvested trees and testing their accuracy to their actual carbon content; this is required for as many species as possible and for the application of both the tree and shrub equations. Another solution is to calculate a national correction using the mature tree equation on shrubs, a modification such as the 9% difference found in this study for the southern Port

Hills and Quail Island areas. Lastly, employing the corresponding equation to the size of each measured tree can also address the carbon difference.

Limitations

This study was limited by the use of a non-specific allometric equation to account for discrete plants. The use of a cuboid equation accounts for empty space as species such as harakeke will not have a dense stem that covers the total area of a cube, thus over-estimating carbon. Harakeke is a common species used in New Zealand restoration plantings, and the need for a more precise equation is evident if this species is to be accounted for. However, in the case of a more mature restoration planting with a closed canopy, species such as harakeke are less likely to be present and the issue will no longer apply.

Conclusion

This is the first study that compares the application of the two allometric equations developed by Beets for New Zealand's native forests on early stages of growth in native restoration plantings. Both equations revealed a higher carbon content the older the planting is. The main equation used in the New Zealand national carbon accounting system (based mainly on mature trees) gave a carbon estimate that was an average 9% higher than that calculated using an equation based on shrub stage trees. The appropriate practice would be to use the allometric equation that applies to the size of the tree (based on the indications of each equation). However, this is more time consuming and the use of two equations and the requirement for more field data collection would need to be balanced against the time available to do the measurements and the costs associated with doing this (the tree equation requires more information to be collected). These results establish the importance of the accuracy of the equation used to calculate the carbon content. Achieving a precise carbon content value translates to best interpretations of possible environmental effects over its sequestration. For further analysis of the data collected in this thesis, both tree and shrub allometric equations were used. The allometric equation that best represented the growth size of the individual tree or shrub measured was applied. This meant that in any one plot, all shrubs and shrub size trees were fitted with the shrub equation, while individuals that had a tree form and tree size were fitted with the tree equation (as recommended by Mark Kimberley, personal communication, February 11th, 2020)

CHAPTER FOUR – Results: Environmental Correlates

Introduction

The planted sites studied include a variety of species compositions and environmental conditions. In this chapter the above-ground carbon present from all plots was analysed to assess whether species composition and environmental factors affected the amount of carbon present. This assessment of the effects of uncontrolled and highly variable restoration sites aimed to identify possible issues that could be addressed to enhance the management of native restoration plantings and increase their carbon content.

The specific goals of this chapter are (1) to understand whether there are any implications of species composition on the total carbon present and (2) whether any environmental factors affect the carbon content of the plantings.

Methods

The amount of carbon present in all plots was calculated from field measurements of tree height, dbh and G (10cm from the ground), using both shrub and tree equations where all individuals were assigned to the equation that met its correspondent dbh size. Discrete plants were summed to each plot's total carbon. These methods are described in detail in Chapter Three.

Atomic Weight

The total carbon calculated by the plot for trees, shrubs and discrete plants was transformed to its correspondent CO_2 equivalence. This value is calculated by the atomic weight of carbon as it represents the total carbon taken from the atmosphere and stored by the individual (equation 8).

$$CO_2 equivalent\ carbon = C \times (\frac{44}{12}) \qquad (8)$$

Environmental Variables

Environmental data were collected in the field to describe and measure plot characteristics. These data were physiography of the terrain (ridge/face/gully/terrace), aspect, slope, slope shape (convex/linear/concave) and presence of mammal pests (deer/sheep/rabbit/mice/stoats/possum) (Table 5). Latitude and longitude were recorded using a GPS with Universal Transverse Mercator coordinates, and the aspect was recorded as degrees North and South facing and were transformed into a percentage of total North or South facing plots to better represent their variability over CO_2 equivalent carbon (Table 6). Biodiversity variables were based on the identification of vascular plant species, their height, diameter and G measured in all the plots and mammal pest records were based on communications with the site manager or owner.

Table 5: Environmental variables recorded for each plot and the methods or tools used to measure them.

Environmental variables measured at the site	**Measuring methods**
Physiography	Visual observation of two individuals
Aspect	Sighting compass
Slope	Clinometer
Slope shape	Visual observation of two individuals
Mammal pests	Information from landowners and managers

<u>Diversity - Species Composition</u>

The Shannon and Simpson indexes were used as the diversity indices. These indexes account for the evenness and abundance of the species present within the plots. These were measured based on the number of individuals of each species present within the plot and that were big enough for the application of the shrub allometric equation (Table 3, Chapter Three).

<u>Species Richness</u>

Woody species richness was measured at a plot level as the total number of woody species present in each plot.

<u>Ordination Analysis</u>

A detrended correspondence analysis (DCA) was carried out using the vegan package in R-Studio, and the resulting ordination scores were plotted in two dimensions.

<u>Data Analysis</u>

Linear models were used to test if environmental variables and species composition could explain variation in the amount of CO_2 equivalent carbon present at a plot level. To further investigate the relationship between the environmental variables and CO_2 equivalent carbon present, a mixed-effects model was used with carbon content as the fixed variable. All variables were classified by factor, continuous or random. Lm, lme and lme4 functions were used from the package lme4, and lmerTest in R to test whether there was any relationship between the environmental variables and the CO_2 equivalent carbon.

Linear regression was done with the total CO_2 equivalent carbon as a constant variable for environmental data such as slope and aspect of the plot towards the North and plot elevation. Further linear model analyses were undertaken using other continuous environmental variables including year of planting, slope, species richness, Shannon and Simpson index. In the case of DCA, the values of the x-axis and y-axis were added in a linear model both with and separate from the environmental factors to see if any significance could be found. Within these linear models the CO_2 equivalent carbon was applied as the constant variable (application of function lm in R). In addition, all factor variables such as physiography, type of grazing present, and slope shape were also included in a separate linear model using CO_2 equivalent carbon as the constant variable (function lm). Both continuous and factor variables were also added together in a linear model using the CO_2 equivalent carbon as a constant (function lm).

Mixed-effects modelling was carried out with all continuous variables using the site of each plot as a random factor (function lmer). More mixed-effects modelling was done: one for all biodiversity variables that were continuous plus the random factor of the site and a second one with all biodiversity variables that were factors plus the random factor of the site (function lmer). Finally, the data was modelled with a fixed effects analysis using two sigmoidal functions i.e. Gompertz equation and Schumacher equation.

Results

The total CO_2 equivalent carbon per hectare present in the plots measured here indicates a possible linear relationship over time (Figure 13). However, it is important to note that due to the lack of planting dates between 1966 and 1990, the amount of CO_2 equivalent carbon being stored between these dates cannot be confirmed.

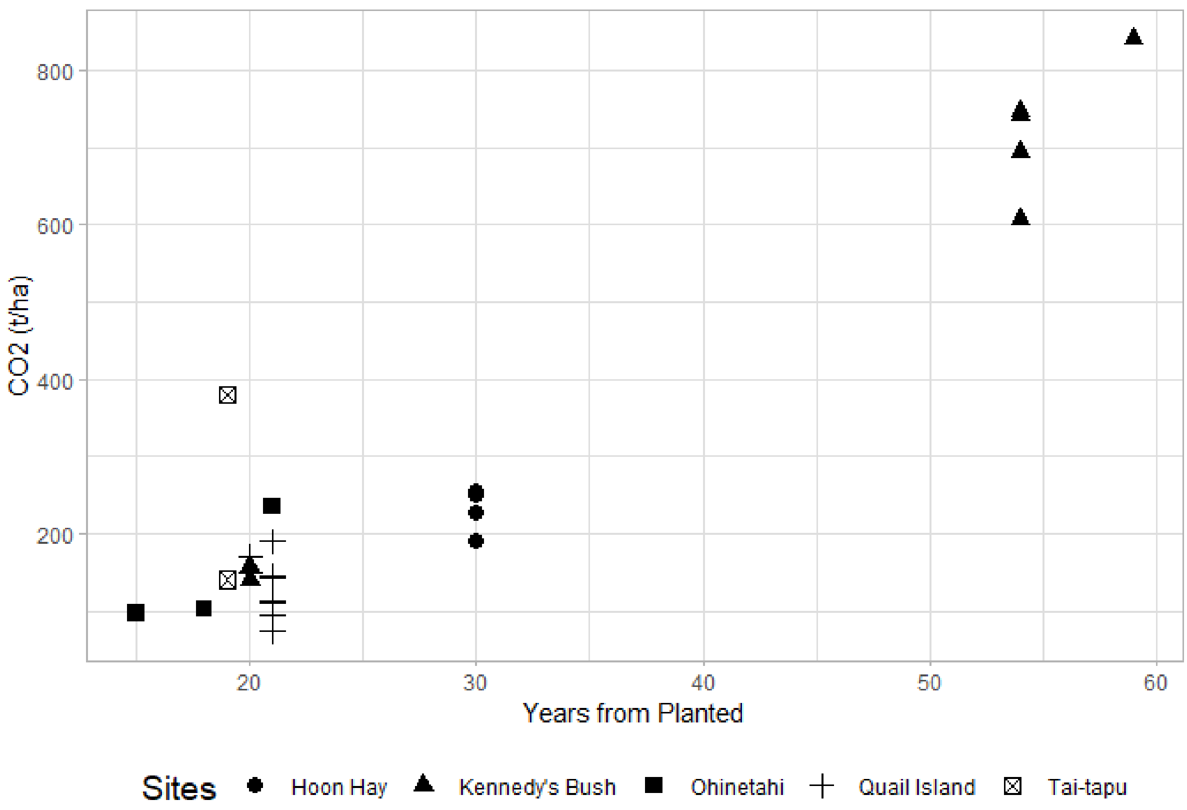

Figure 13: Total CO_2 equivalent carbon of all 25 measured plots and their age since planted.

<u>Environmental variables</u>

When comparing the environmental variables recorded on-site (aspect, elevation, slope and physiography), to the total tonnes of CO_2 equivalent carbon present per hectare within three planting age groups, no clear relationship is apparent (Table 6). The amount of CO_2 equivalent carbon present is clearly higher with age of the plantings. As the plantings grow they increase in biomass, indicating that until year 59 the plantings have grown and sequestered carbon irrespective of the local environmental conditions. With considerable variability amongst the plantings and the low number of sites and plots measured to represent these, it is impossible to see any clear link between the environment and the amount of CO_2 equivalent carbon that has been sequestered.

Table 6: Environmental variables and CO_2 equivalent carbon for sites with a close year of planting.

Year of planting (n=number of plots)	Total CO_2 equivalent carbon (t/ha)	Elevation Mean (m.a.s.l.) and Standard Deviation	Aspect (%)	Physiography	Slope Mean (%) and Standard Deviation
1999-2005 (n=16)	152.7441667	138.625 +/- 33.5	N=100% S=0%	Face=75% Ridge=12.5% Terrace=6.25% Valley=6.25%	29.625 +/- 3.2
1990 (n=4)	231.1283	438 +/- 10.2	N=75% S=25%	Face=100%	17.75 +/- 5.8
1961-1966 (n=5)	725.9193333	472.2 +/- 7.8	N=50% S=50%	Face=100%	47.8 +/- 6

<u>Linear regression and mixed-effects model</u>

Results from linear regression and a mixed-effects model showed no clear relation between the amount of CO_2 equivalent carbon for each plot and their biodiversity and environmental conditions. However, a tree model showed the year of planting, and the slope angle can affect the CO_2 equivalent carbon when this is above 15.5 degrees (Figure 14). The tree model shows a possible effect over carbon content for plantings older than 42 years since planted and if the slope is above 15.5 degrees then the CO_2 equivalent carbon will be less. When the year of planting and slope variables were tested on linear regression, no significant results were shown.

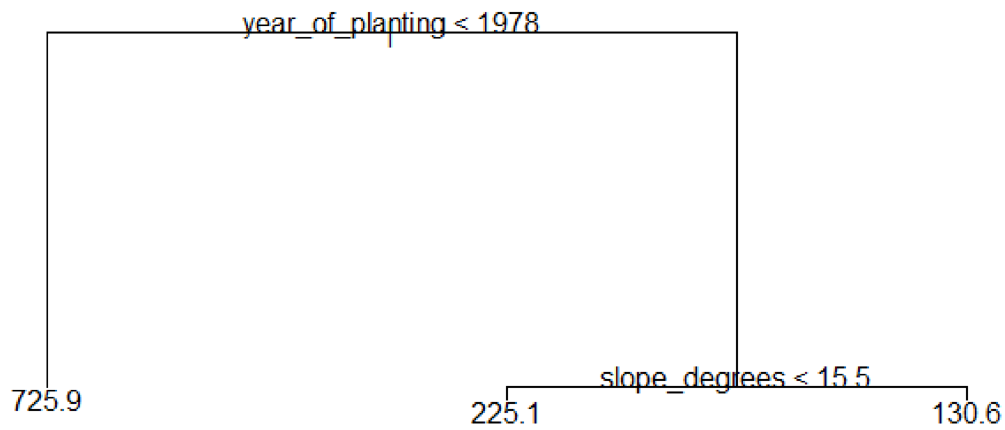

Figure 14: Tree model showing a possible effect of CO_2 equivalent carbon by the age of planting and when the slope is less than 15.5 degrees.

A fixed-effects analysis was undertaken with the CO_2 equivalent carbon of all 25 plots. The model used was a sigmoidal curve where a Golpertz and a Schumacher equation were fitted for comparison. Results were not promising, and residuals were unbalanced due to the low number of plots and high variability—all attempts to apply a model to the data resulted in inconclusive outcomes.

Discussion

There was a clear increase in carbon content over time since planting (Figures 13, 14 and 15). The older the plantings are the more CO_2 equivalent carbon they have. The CO_2 equivalent carbon does vary from at a plot level due to different environmental factors. A tree model (Figure 14) which indicated that age and slope could impact the carbon content was carried out. In this study, it is evident that the older the planting date, the more carbon was present. However, analysing a linear regression and a mixed-effects model slope showed no significance over the carbon contents, leaving inconclusive results. Environmental factors, at least across the sites studied here, do not however appear to have any significant effect on the CO_2 equivalent carbon present in biomass. Having a control study that oversees the effects of slope over carbon content can inform us better of a possible influence.

Limitations

The lack of restoration sites 31 to 53 years old makes it difficult to state with confidence that carbon contents across the first 60 years of restoration plantings increase in a linear manner. However, the CO_2 equivalent carbon content values in the 53+ year-old plantings clearly show that carbon continues to be sequestered in older restoration stands.

Repeated measurements of the same plots could indicate how they sequester carbon over time independently from each other. Because of a lack of time constraints of this study (primarily due to the Covid-19 lockdown), it was not possible to obtain more data for this purpose.

Due to the high variability and the low number of plots across a wide range of years, no pattern was observed between the environmental variables and the carbon contents in biomass. This does not rule out that a specific variable may have a direct effect (e.g. slope or elevation). To understand this, an in-depth study of each variable should be performed as a controlled study.

Conclusion

There are no clear indications from this study that a specific environmental condition or species composition may have a direct effect on the carbon contents in restoration plantings in the southern Port Hills and Quail Island. However, there is a clear pattern showing higher carbon contents with older restoration plantings.

CHAPTER FIVE – Results: Comparisons with the MPI Look-up Table.

Introduction

After gaining a better understanding of the allometric equations used and the effects of species composition and environmental factors on carbon present in plots from previous chapters, this chapter provides a comparison of the CO_2 equivalent carbon measured in this study and the government's official estimates of carbon stores present at particular ages using the MPI look-up tables. As the official method used to calculate post-1989 forest carbon stocks in areas less than 100 hectares is a set table of values per year with no region, management effort or species differentiation for native species (MAF, 2009), it is considered useful to compare these with real data such as that collected in this study. These tables are called the look-up tables, and they are an online government resource created under the 'Climate Change (Forestry Sector) Regulations 2008' (NZGovernment, 2008). The carbon sequestration values within these tables for indigenous forests aim to cover the first 50 years of native forest carbon storage in a regenerating or planted (restored) native forest. The values were set by the Ministry of Agriculture and Forestry (MAF), an historic government agency that is now part of the Ministry for Primary Industries (MPI) and are based on research undertaken by Landcare Research. The data used by Landcare Research consisted of 52 sites of retired grasslands converting into natural regenerating land between North Auckland to Otago, and 42 of these sites constitute predominantly of regenerating Mānuka and/or Kānuka. The other 10 sites consisted of tree ferns, *Coprosma* species, *Ulex europaeus*, *Cytisus scoparius,* mahoe, kowhai, and *Pseudopanax arboreus.*

In this chapter, I compared the carbon present at particular times as estimated using the look-up tables with the carbon measured in my study using science-based New Zealand allometric equations for native species within a restoration context. This allowed comparison of the fixed carbon yields in the look-up tables with different restoration planting sites where randomised plots of high floristic variability were measured. In addition, the values from this study were compared to those used on the only written report released by Landcare which has been used as one of the sources the look-up tables were based upon (Payton et al., 2009).

The goal of this chapter is to compare the CO_2 equivalents carbon expressed in the look-up tables (Schedule 6 table 2 column 5) with the values calculated using the allometric equations (Chapter 3) for my restoration study sites of the southern Port hills and Quail Island.

Methods

The CO_2 equivalent values applied in Chapter Four based on the allometric equations of Chapter Three were used for the comparison to the look-up tables. The government values were taken from the current MPI's indigenous forest look-up table.

Results

The carbon values of the MPI look-up tables include carbon pools that were not measured or included in the 25 plots of this study. These pools are coarse woody debris (CWD), roots and fine litter on the floor. Despite these differences, comparison of t/ha of $C0_2$ equivalent carbon of all 25 plots by year of planting with the values from the MPI look-up tables (Figure 14), shows that most of the plantings of 30 years or less since planting fall within the look-up tables range. With only one outlier value surpassing the highest amount predicted from the look-up table we have a reasonably strong relationship. The outlier value which represents the bottom plot of the Tai-Tapu site is quite an important indication that higher carbon content can be achieved under optimal conditions.

The oldest planting sites of Kennedys Bush Reserve have a $C0_2$ equivalent carbon value above the values of the look-up tables, although the tables only extend to 50 years (Figure 14). These two sites are 54 and 59 years, and the results suggest that carbon content continues to rise with time as opposed to flattening out as the look-up tables imply. On comparison of this study's data with that of the Landcare Research study (used to extrapolate the MPI look-up table) we get a clearer understanding that past 50 years of age, the CO_2 equivalent data that they included for planted and natural regenerating forests are below those that I recorded (Figure 15). The carbon values of the Landcare Research study also included pools that this study did not measure, meaning that the difference is likely to be even larger.

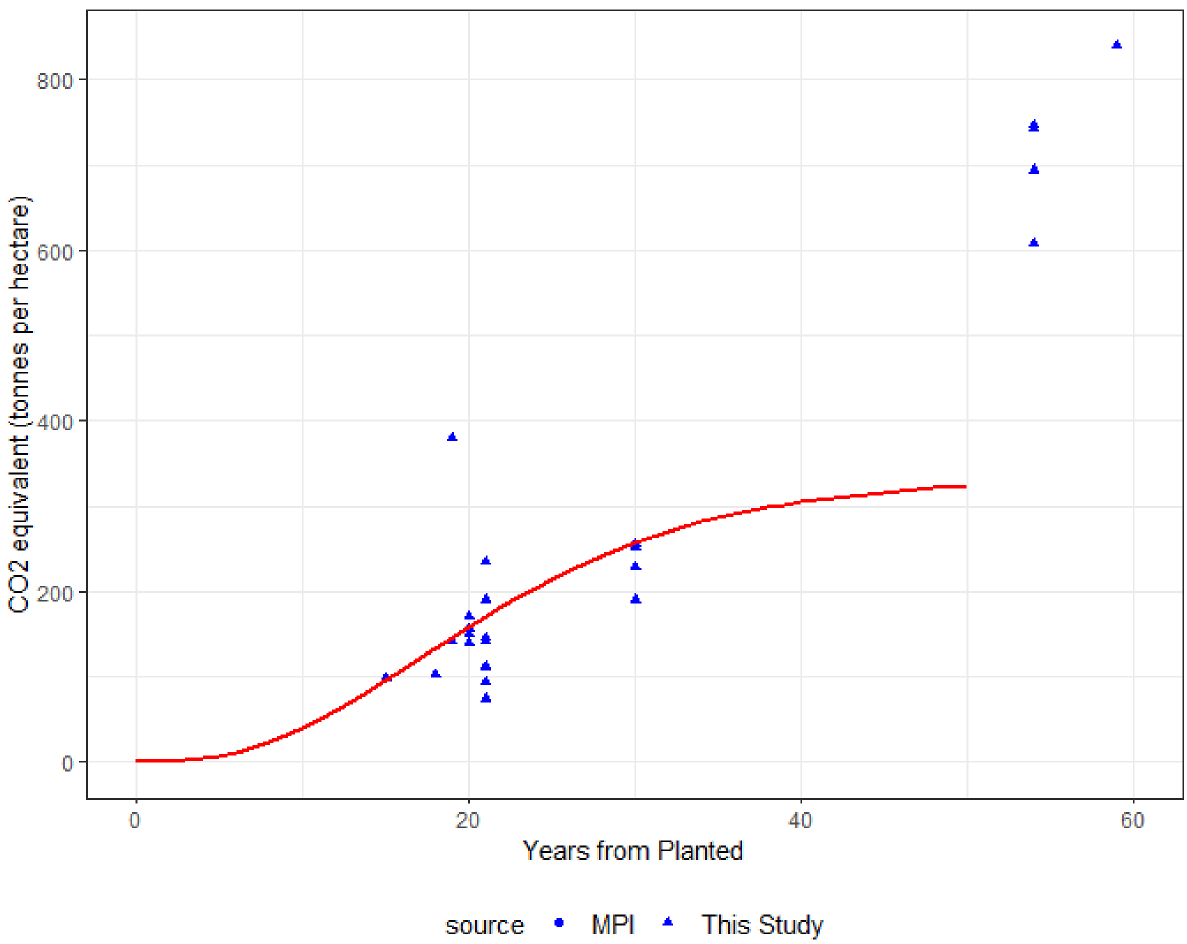

Figure 15: Total CO_2 equivalent carbon by the year of planting. Blue triangles for the carbon sequestration of the 25 plots measured in this study and a red line for the MPI look-up table values post 2010.

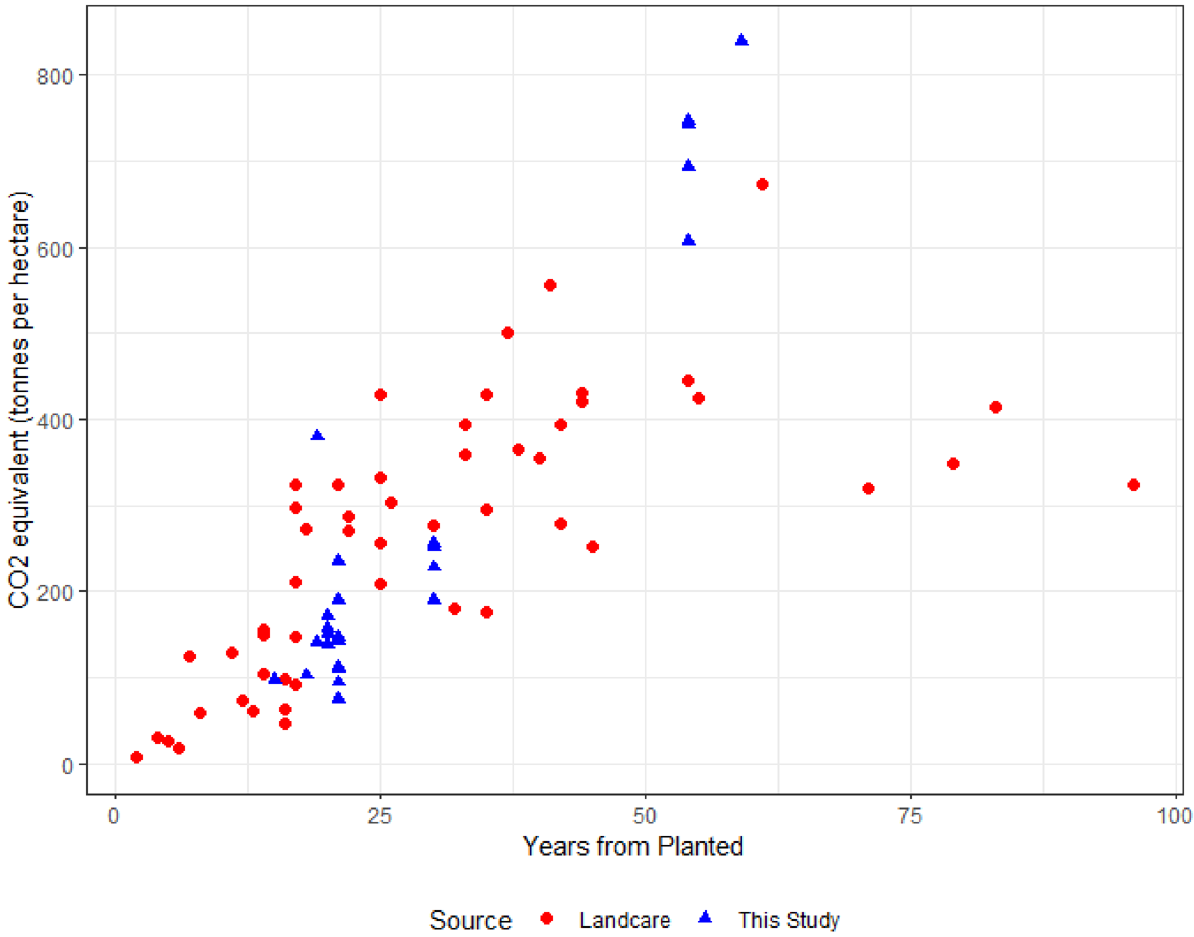

Figure 16: Total CO_2 equivalent by the age of planting. Blue triangles for the carbon content present in the 25 plots measured in this study and red circles for the 52 sites measured by Landcare Research (Payton et al., 2009).

<u>Discussion</u>

With the exception of one outlier, CO_2 equivalent carbon amounts in biomass were close to the CO_2 equivalent carbon expected from the MPI look-up tables for native regenerating shrublands up to the age of 30 years. The outlier (bottom plot of Tai-Tapu) contained more than double the amount of CO_2 equivalent carbon than that expected by the look-up tables. This was likely caused by the site physiography, being the only plot located on a lower valley slope. Physiographic characteristics are known to present a multi-scale influence over tree growth (Littell et al., 2008; Lo et al., 2010; Martin et al., 2007). The effects can be important as it combines climate and nutrient interactions that influence tree growth (Hawkins et al., 2010). The effects are species dependent as physiography impacts can differ by each species' particular needs and, consequently, their carbon sequestration uptake (Rieger et al., 2017). All species present in the outlier site had a high dbh, G and height with *Myoporum laetum* being the most common species. Another characteristic that has a relationship to carbon content is the moisture content of a site. High moisture content within the limits of >200mm per annum relates to increased carbon content (Payton et al., 2009). Due to the physiographic conditions of the valley in the outlier site, a moisture content >200mm per annum is expected; this is the ideal water availability for carbon sequestration. More plots and sites planted within a valley for a controlled study could help understand this significant increase in carbon content in this topographical position and moisture content implications.

This study was restricted to above-ground carbon (AGC) by the fact that few studies have assessed the effects of below-ground carbon (BGC) in New Zealand's native trees. Because of this, most researchers will add between 20 to 25% of the total AGC to their measurements to account for the root carbon content as per the IPCC standards and a few results that have shown to be closer to a 20% BGC in New Zealand (Beets et al., 2012). The look-up tables do not specify how the root carbon was calculated or what percentage was used to do so. This additional carbon is insufficiently explored and explained within the current literature and needs further study. Root carbon and soil carbon pools were not included in the 25 plots of this study. Other carbon pools that weren't included are floor litter and CWD. The carbon values expressed in the look-up tables did include these carbon pools, a meaningful difference that must be considered when comparing their carbon values with this study (Figure 15 and 16). However, that I did not include these additional carbon pools (BGC, CWD, litter) suggests that at least for my study sites, the look-up tables underestimates the actual carbon present across all ages but especially for older plantings.

Another important difference is that the look-up table only presents carbon content values up to 50 years of native regenerating shrublands, while in this study I measured the carbon contents of plantings up to 59 years old. The plantings beyond 50 years appear to have higher carbon contents than the look-up tables imply. Compared to that of the look-up table, the carbon contents of this study did not flatten between years 30 and 50. In addition, comparing the results from this study to the Landcare Research study (used to create the look-up tables), we distinctly see an increase of carbon contents between 30 and 53 years from planting. As the Landcare Research study included 2 to 96-year-old plantings sites, both studies' carbon content amounts diverged after 30 years. Between thirty and fifty-four years from planting, native mixed forests increase their CO_2 equivalent carbon content compared to regenerating shrublands with a predominance of Kānuka and Mānuka species (Trotter et al., 2005). This gain suggests that after 30 years of age, restoration plantings may continue to sequester carbon at a higher rate than regenerating shrublands. The increase may be due to the different species and management of both. We must be mindful that the look-up tables and the Landcare Research study were based exclusively on regenerating shrublands dominated by species of Kānuka and Mānuka between North Auckland and Otago (Trotter et al., 2005), and included more carbon pools than this study. On the other hand, community plantings increase biodiversity and have a more diverse species composition.

The results of this study indicate a continuation of increased carbon content for restored forests. This suggests the need for further research on the carbon sequestration life cycle of native species and their management.

Limitations

Moisture content and rainfall was not considered in this study. However, other literature has found such a relationship and may clarify increases and decreases in this study's sites.

Conclusion

The CO_2 equivalent carbon content values from the look-up tables based on native regenerating shrublands match best with the data gathered for this study for younger sites, except for one outlier that likely reflects a local site effect. Despite the different carbon pools, sites and establishment, the data presented here also suggest that at least up to the 30-year-old plantings, most restoration projects in the southern Port Hills and Quail Island contained CO_2 equivalent carbon in similar or greater amounts to that of the MPI look-up values and the Landcare Research study. In older sites, my measured carbon contents are substantially greater than the look-up tables imply. Different management and species composition between the data collected here and that used to derive the look-up table may be the cause of this.

Plantings containing higher CO_2 equivalent carbon in their biomass than indicated by the look-up table at older ages emphasise the need to continuously gather data to better understand the effects of carbon content on species' composition and environmental conditions at the sites where we plant and restore for biodiversity. A proficient study that addresses the carbon content involving a national scale data collection of restoration sites with known planting dates will allow a sensible model and carbon table to be created and used for restoration plantings around the country. Keeping a clear record of these plantings and protecting them in perpetuity will also help us focus current efforts towards reaching a more efficient 2050 carbon-neutral goal.

CHAPTER SIX – Results: Carbon Content of Individual Tōtara Plants

Introduction

Efforts to reforest degraded land by planting indigenous trees have provided important habitat for indigenous biodiversity. These growing habitats allow vulnerable and threatened species to thrive with proper management within highly modified or degraded landscapes (Chazdon & Brancalion, 2019; Norton et al., 2018). Previous chapters have highlighted that these plantings are also important for sequestering carbon. The longer these plantings have been established, and so long as degrading factors are minimised or eliminated, the more biodiversity and carbon they hold (Carswell et al., 2012). One feature of many restoration plantings and regenerating native forests is that they tend to be dominated by early successional tree species and lack later successional species (Forbes et al., 2020). Enhancing these areas with other tree species can further improve both their biodiversity values and their efficiency at carbon uptake (Forbes et al., 2020). Tōtara is a long-lived canopy dominant species that is moderately light-demanding and is a dominant canopy tree in many mature native forests, especially in this study area. Tōtara can survive if planted at the very start of the restoration process (Bergin, 2003; Ebbett, 1998) and existing planted tōtara around the Port Hills and Quail Island highlight that we can save time by planting this long-lived species from the start of the restoration. Planting trees like tōtara can potentially sequester more carbon than would be the case with mixed plantings lacking this and other later successional tree species (Williams & Norton, 2012).

Notwithstanding this, plantings of tōtara are scarce in the southern Port Hills and Quail Island, and interest in planting more of this species is high, including in new plantings and as an enrichment species (Forbes et al., 2020) into existing plantings and areas of natural forest regeneration. Understanding how much carbon content individual tōtara trees sequester can allow us to better assess their potential for offsetting CO_2 emissions. As more individuals are planted further data will become available for future use, improving carbon sequestration estimates for tōtara in the southern Port Hills area. In this chapter, I present the data on CO_2 equivalence in the biomass of 105 planted *Podocarpus totara* individuals.

Methodology

A review was first undertaken to identify the areas where tōtara were planted in the southern Port Hills and Quail Island, and to obtain the dates of planting. Quail Island was automatically excluded as the tōtara found there were not of shrub or tree size yet. Confirming the planting dates of the recorded trees with historical images was necessary as all tōtara found growing within grass areas pre-dated the five restoration projects as these were surviving individuals from previous land clearing events and were excluded from this study.

Tōtara measurement

105 tōtara trees were growing amongst mixed species plantings. These were all restoration planting sites, no single-species tōtara stands were present and open grown tōtara trees were excluded. Trees were found growing within restoration plantings amongst other tree species, on the edge of these restoration plantings and amongst gorse. Every individual identified as a planted tōtara species growing within these sites was measured. An estimation of the year of planting was obtained for all individuals included in the study (based on discussions with people who knew the sites). Individual measurements were taken within the plot and its surroundings following the same protocol as was applied to all other trees, shrubs and saplings within the 10x10m plots (Chapter 2). They were recorded in separate sheets and identified as a single species within the restoration plantings (RP), on the edge (E) of the restoration planting within a 2m distance from it, and growing amongst invasive gorse (G). Single standing individuals usually amongst grass species were not recorded as they were found to usually predate the restorations.

Species wood density

The species wood density used to calculate the carbon content for all tōtara measured is 446 kg/m^3 and is based on a sample size of four-hundred tōtara trees at a maximum age of fourteen years (Bergin et al., 2008). The Landcare Research database was not used as its sample size is two and no further information regarding the method or age of the trees is provided.

Carbon content

The total carbon content was calculated as atomic weight (see methodology of Chapter 3 and 4). To obtain the mean yearly carbon accumulation rate the total carbon content was divided by the tree age (Beets, Kimberley, Paul, et al., 2014; Trotter et al., 2005).

Data analysis

Three separate mixed-effects modelling was undertaken using the same sample of the data. An ANOVA was then carried out to look for any significant differences in the content of CO_2 equivalence carbon amongst the different growing conditions of tōtara found growing in the study sites. Due to the multiple analysis over the same data, a Bonferonni correction was calculated to check that the significance had no added errors from the multiple analysis.

Results

The total amount of CO_2 equivalent carbon in planted tōtara ranged from 1.07 kg per tree for a nineteen-year-old individual to 896.94 kg per tree for a thirty-year-old individual. The range in age and conditions of growth were recorded and are comparable between the height and the dbh of the individuals (Figure 16), as these measurements are directly related to the carbon content of any tree (Wang & Gao, 2020). A linear relationship between the dbh and the height is present.

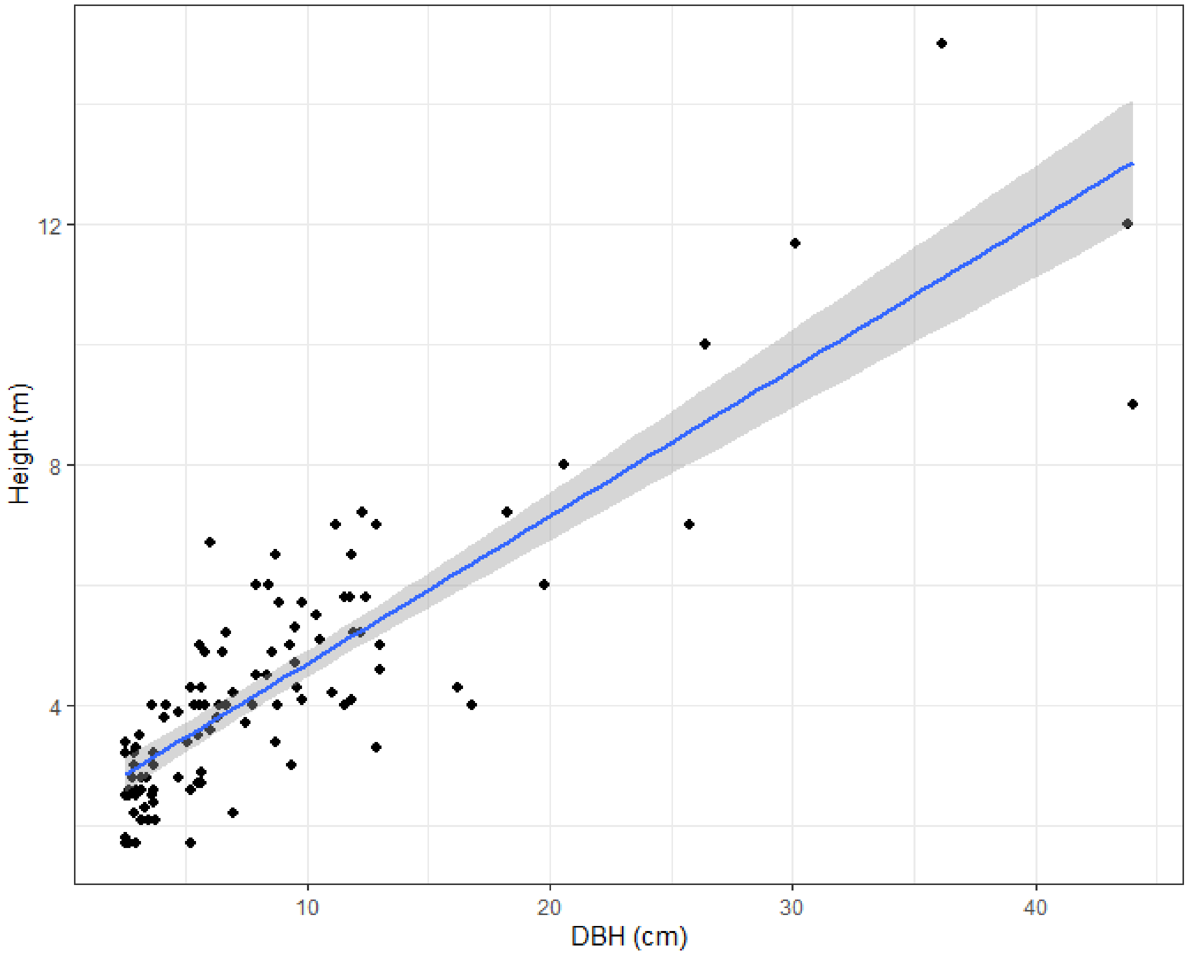

Figure 17: Height and dbh of all tōtara measured in the southern Port Hills and Quail Island.

When comparing the different growing conditions for all planted tōtara in relation to the CO_2 equivalent carbon in tonnes per tree and the dbh we can see the curvy-linear pattern persists in all of them (Figure 17). Due to a lack of available tōtara in all growing conditions, only the plantings from 2001 can be easily compared in terms of carbon contents (Figure 18). From this year, planted tōtara growing on the edge of the restoration sites sequestered higher amounts of carbon in the first nineteen years from planting than at the other two site types.

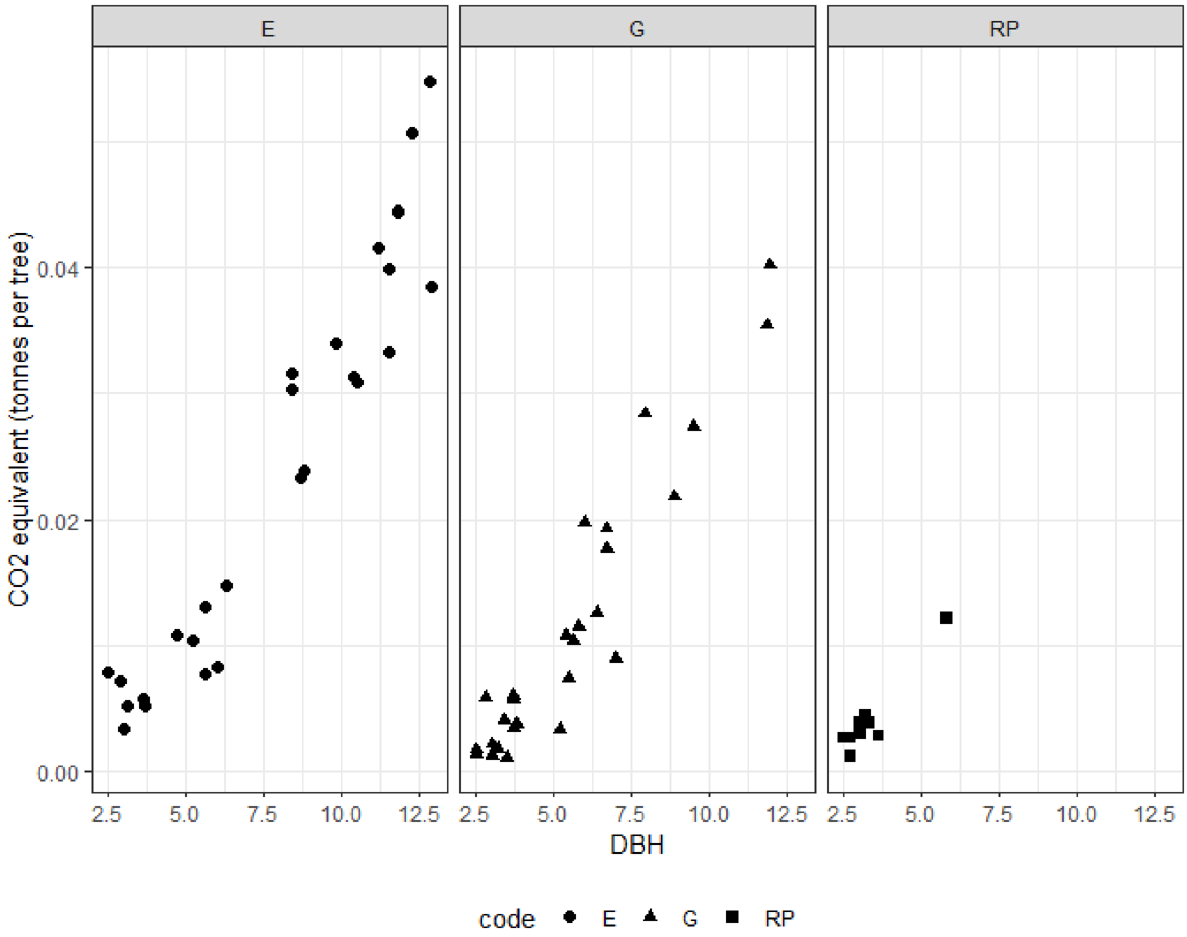

Figure 18: CO_2 equivalent (tonnes per tree) by the dbh under different growth conditions for planted tōtara: restoration plantings (RP), edge (E), and gorse (G).

To further understand what is happening with the carbon content of planted tōtara all sites are compared by age and growing conditions (Figure 19). In this case there is a clear indication of higher carbon content the older the trees are. A scaled power transformation is applied to all ages and CO_2 equivalent amounts to better represent the data (Karian & Dudewicz, 1999). When looking at it on a graph (Figure 20) the data was better distributed and a clear pattern was visible, where the best growing tōtara of all ages increased their carbon content with age and can contain more than 0.5 tonnes of CO_2 equivalence at 30 years. This, however, also shows a high variability in growing conditions between tōtara trees of the same age, reflecting that growing conditions are affecting their carbon content.

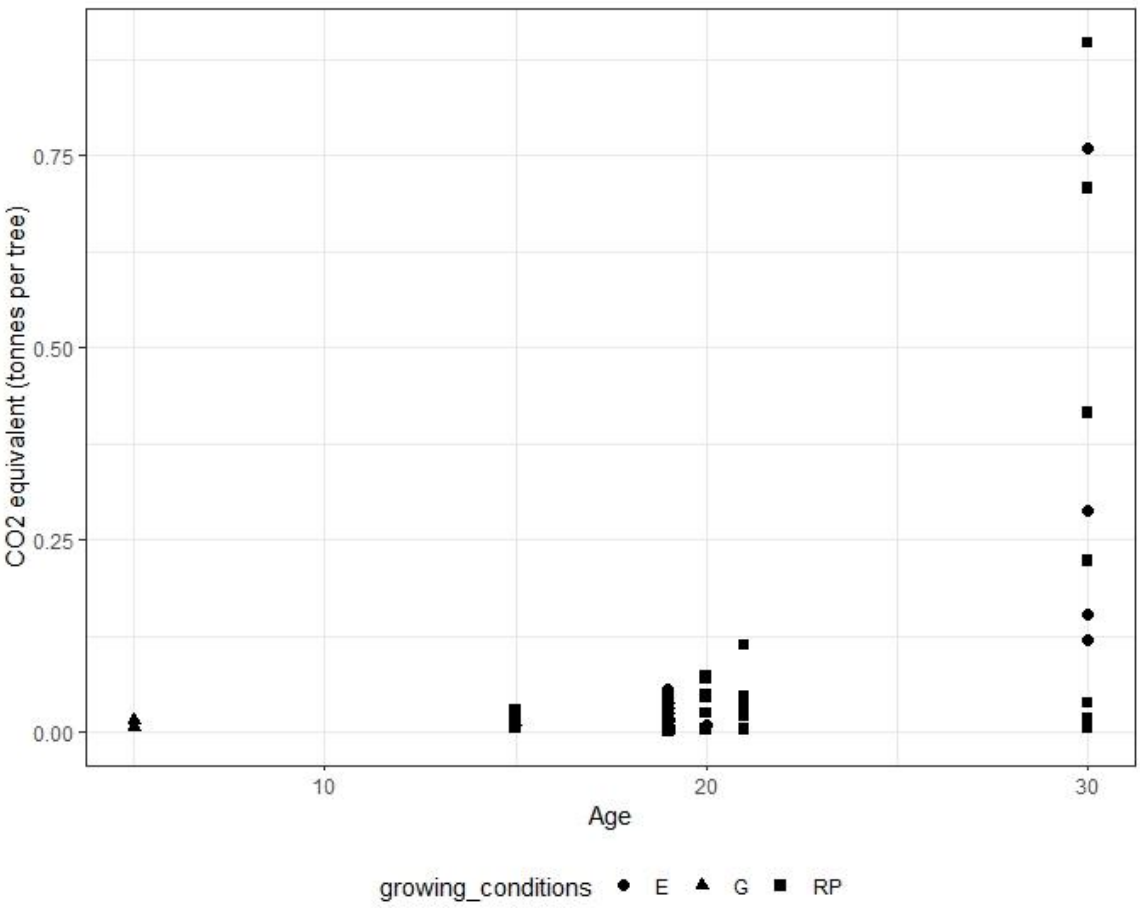

Figure 19: CO_2 equivalent (tonnes per tree) by the age of the tree and growing conditions (RP=restoration plantings, E=edge and G=gorse) of tōtara trees.

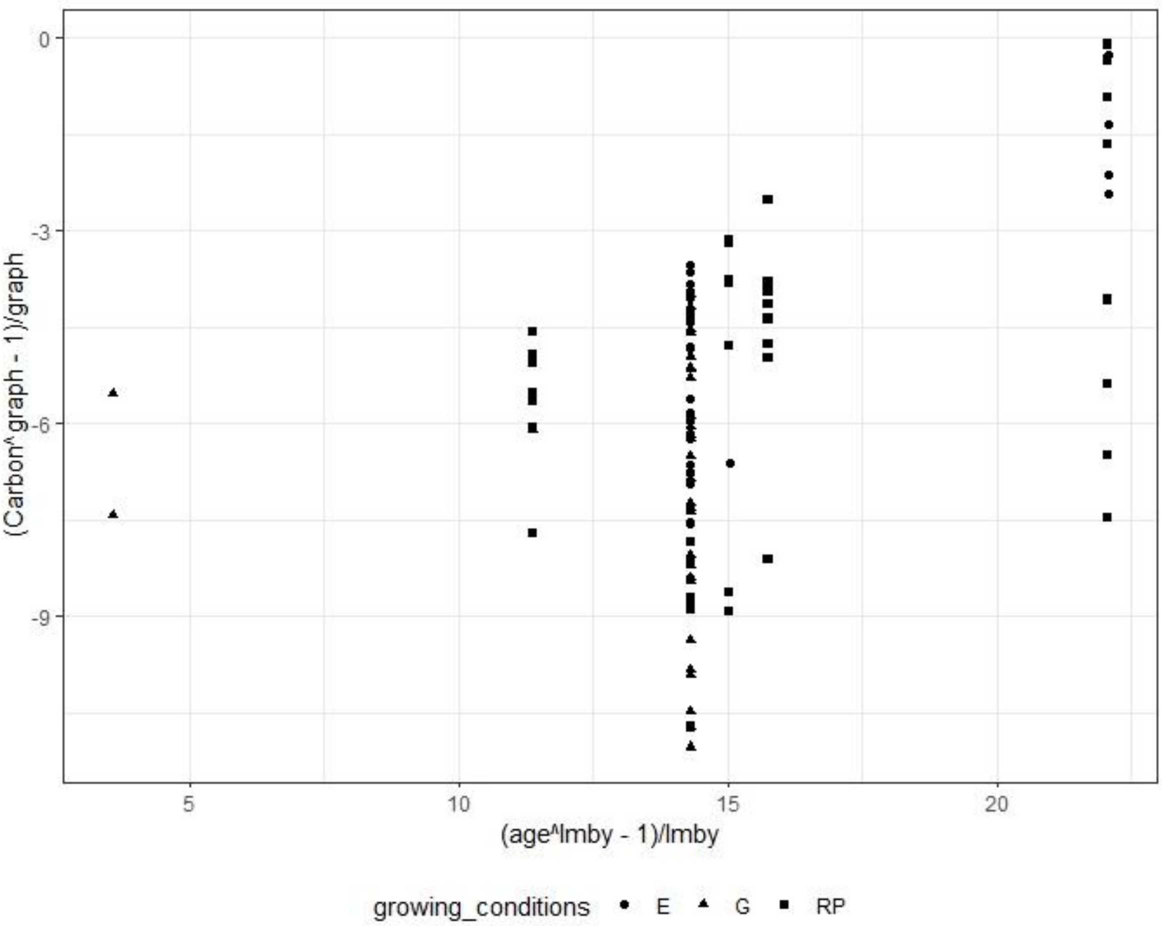

Figure 20: Lambda value of CO_2 equivalent (tonnes per tree) by the lambda value of age of the tree and growing conditions (RP=restoration plantings, E=edge and G=gorse) of tōtara trees.

A mixed-effects model was undertaken using the different sites as the constant variable, the carbon contents as dependent variables and growing conditions as fixed effects. Results showed differences in p-values for all growing conditions (around gorse, within the restoration plantings and by their edge). To see what conditions were significant, these were separately tested in an ANOVA type III test. The results showed that the carbon context of totara at the restoration planting's edge was significantly greater than at the other two site types (Table 7). The Bonferonni correction was calculated to eliminate any additional error to the p-values resulting from the multiple analysis done to the same sample of the data. The Bonferonni correction confirmed that tōtara planted on the edge of a restored forest had significantly higher carbon contents.

Table 7: Results of three mixed-effects models of the different growing conditions, age and Bonferonni correction.

Fixed variables + random factor (site)	**p-value**
(Intercept)	1.844e-06 ***
Growing conditions (RP and G)	0.1419
Age	0.0255
Fixed variables + random factor (site)	**p-value**
(Intercept)	9.479e-05 ***
Growing conditions (E and RP)	8.622e-07 ***
Age	0.07778
Fixed variables + random factor (site)	**p-value**
(Intercept)	0.0053503 **
Growing conditions (E and G)	0.0002783 ***
Age	0.3022731
Bonferroni correction [1-(1-p value)^1/amount of tests**]**	**0.017**

Discussion

The data and results must be interpreted with caution due to the low number of samples and high variability between the sites and growing conditions. This study was carried out to help guide new tōtara plantings for carbon sequestration that are proposed in the area. Further research on new plantings of tōtara should include the species' survival rates to account for the full effects of different sites and growing conditions. Other research has indicated that the survival of planted tōtara can be restricted by browsing ungulates (Forbes et al., 2016). The susceptibility of this increases in edge trees as they are more exposed to browsers like deer (Tulod et al., 2019). Proper management is required for consistently high carbon content from tōtara plantings. Plantings should be done preferably on the edge of forests where browsing by deer and other ungulates is low or non-existent.

Research on the closely related *Podocarpus laetus* suggest an annual carbon gain of 0.1 and 0.4 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$ for 250 and 1,000 stems per hectare natural stands can occur (Williams, 2010). The mean rate of carbon sequestration of *Podocarpus totara* growing in restoration plantings of mixed-species measured in this study ranged between 0.5 and 2.1 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$ for 250 and 1,000 stems per hectare, respectively. The *Podocarpus totara* sites studied here involved measurements of single trees growing at lower elevations than those where Williams (2019) studied *Podocarpus laetus*. Despite the site differences, measurement and planting management, this comparison shows that *Podocarpus totara* can potentially sequester higher amounts of carbon than *Podocarpus laetus*.

Growing conditions of both Podocarpus studies differed due to stand size, site elevation and site conditions (Port Hills and the South Island high country). This study measured tōtara growing in restoration planting where not all areas had a closed canopy and many individuals were found growing around gorse. The *Podocarpus laetus* study was based in naturally established closed-canopy stands where trees competed for light due to stand size restrictions, impacting the carbon gains. These restrictions would have predominantly been the case for the 1,000 stems per hectare where the competition for light would have limited their growth. Furthermore, carbon gains may also be affected by the variation in dbh between the two Podocarpus. *Podocarpus totara* registered a mean dbh of 8.8 cm and mean height of 4.4 m at a year range of 5 to 30 years and *Podocarpus laetus* is predicted to have a mean dbh of 14.3 cm and a mean height of 5.2 m at 100 years.

Limitations

The sample size and amount of variability is a limitation as this may introduce bias to the results. These results are only indicative for future plantings. The survival of planted tōtara is unknown and historical images do not provide enough information for further assessing this. One measurement of these trees does not provide a robust assessment of the annual carbon increment of this species in this study site, but it is indicative for comparison purposes only. Nonetheless, the results presented here do show that an individual planted tōtara in ideal conditions can store in excess of 0.5 tonnes of CO_2 equivalent carbon at 30 years.

Conclusion

Podocarpus totara is a long-lived species that may sequester higher amounts of CO_2 equivalent carbon in the first few years of restoration projects if planted at the edge of a forest and provided ungulate control is possible. This is a fast growing species that sequesters a considerable amount of carbon for which it is important to be included in the beginning of restoration plantings or as an enhancing species. As this species grows and becomes older they sequester more carbon and their care and maintenance becomes crucial for their permanency. With good management the survival of tōtara as an enhancing species for carbon sequestration purposes can be achieved. Perusing the enrichment of planted restoration areas and future plantings with tōtara is important to achieve higher carbon content in them. From a restoration planning perspective, the results of this study do suggest that an individual planted tōtara under ideal conditions can store in excess of 0.5 tonnes of CO_2 equivalence carbon at 30 years (excluding below ground carbon associated with these trees).

CHAPTER SEVEN – Discussion, Limitations, Assumptions and Conclusion

Discussion

The carbon content of restoration plantings was calculated based on tree allometric equations. Two of them were compared to see how the national use of one of them for all trees and shrubs is representing the carbon contents in the national accounting system that MfE reports to the UN. Once these differences were understood, the carbon contents were transformed into their atomic weight at a plot level to establish if the carbon amounts measured may be due to any site variability. These random samples were then compared to the national estimates for carbon contents at different ages that the Ministry of Primary Industries uses to calculate and pay carbon credits. The data collected here were also compared with one of the studies used to create the official carbon look-up tables. Finally, the CO_2 equivalent carbon values for *Podocarpus totara* individuals found in the study sites were then analysed to explore their potential as enhancing species for carbon sequestration purposes.

Allometric Equations

Shrub and tree allometric equations were compared using the same individuals, regardless of their size and best applicable equation. Except for one outlier, results showed that plots of similar age had similar carbon contents. Further data analysis also showed that the older the plantings, the more carbon they contained with no decline observed up to the oldest plots measured here (59 years).

Overall, the use of a single allometric equation for all trees and shrubs from plantings that were between 15 to 59 years old showed that the mature tree equation resulted in CO_2 equivalent carbon estimates that were 9% higher than when a shrub-based equation was used. As the national accounting system uses the mature tree allometric equation, this difference can be significant for younger stands that lack mature trees. The measurements needed for both equations are labour-intensive, costly and increase observer bias (Kapfer et al., 2017; Milberg et al., 2008). Using a single equation and adjusting the percentile difference could be better than altering the data collection methods at a national level to include both methods. If no use is made of both equations, ground truthing is necessary to understand each equation's application. For smaller research samples and data collection, the use of both allometric equations is optimal.

Environmental correlates

The amount of carbon sequestered in my study plots did not show a significant relationship with different combinations of species composition and environmental variables (Table 6). A single outlier plot (bottom plot in Tai-Tapu) is located at a site with different topography, the only plot located in a valley at the bottom of a basin. This site is atypical for restoration because it is a more productive location which is likely to have resulted in better tree growth (Bueis et al., 2017; Omary, 2011; Payton et al., 2009; Smith et al., 2017). Due to the lack of data on similar sites and variables, the relationship could not be further examined.

Comparisons with the MPI look-up table

The MPI look-up tables were initially based on studies that investigated long-term average net increase of CO_2 equivalent carbon contents from native species at a national scale and resulted in an annual rise of 3 t CO_2 h^{-1}. This original look-up table did not present any variation on the carbon uptake with age, being based on a constant rise. These values were updated in 2010, where new values doubled the carbon contents over 50 years (MAF, 2009). These values were not constant as they were fitted using a Gompertz equation (Payton et al., 2009), giving a lower increase in carbon content from about 30 to 50 years from planted (Figure 15). The current values were based on regenerating shrublands around the country, where the dominant species are of the genus *Kunzea*. The generalization of carbon content for all native afforestation of New Zealand's forests based only with ones that have a main composition of Kānuka and Mānuka was justified by the fact that this type of native cover accounts for nearly 70% of the total regenerating indigenous afforestation of New Zealand (Carver & Kerr, 2017).

Even though the values of the native forests look-up tables were considered to be too high for indigenous regenerating shrublands (MAF, 2011), my results from restoration plantings in the southern Port Hills and Quail Island (which excluded below ground and CWD carbon pools) show that measured values in plantings and the current look-up tables for indigenous forests are if anything an underestimate of actual carbon sequestration,. However, my study also showed the need to extend the look-up tables to beyond 50 years as my results show that carbon content continues to increase after 30 years, in contrast to what the look-up tables imply (MAF, 2011). While a single table for all indigenous forests is too broad, the current values represent restoration planting initiatives from my study's sites up until year 30. After that, carbon content values in the look-up tables need to be revised, as in my study plantings older than 30 years contained more carbon than the MPI look-up tables suggest. These tables also need to include site differentiation for New Zealand native species based on regions, as is done with radiata pine. It is likely that carbon sequestration will be greater in other parts of the country such as Northland or Gisborne, as the look-up tables imply for radiate pine (MPI, 2017). Canterbury is known to have lower productivity and therefore slower growth rates. This low productivity of restoration plantings in Canterbury might not be replicated in other New Zealand areas where higher carbon sequestrations could be achieved. The representation of natural regeneration and plantation forest of native species may well require revision as well; they do not have the same management, species composition or planting densities. For instance, in some parts of New Zealand, Kauri plantings that are managed for wood production grow faster than naturally regenerating stands (Steward & McKinley, 2005). These aspects are not accounted for in a single look-up table.

The mean annual carbon content of the measured restoration plantings of this study is 2.4 $t{\cdot}ha^{-1}{\cdot}yr^{-1} \pm$ 0.4 well above the expected national mean of native carbon sequestration rates for native forests of 1.4 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$ (Kirschbaum et al., 2009). It is closer to the mean annual carbon sequestration for the first 40 years of regenerating shrublands dominated by Kānuka/Mānuka which range between 1.9 to 2.5 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$ (Trotter et al., 2005).

Tōtara

Using tōtara for enrichment planting to existing restoration plantings can restore future canopy dominant species, improve biodiversity, and add to the above-ground carbon content pool (Forbes et al., 2020). With good management, this species might perform well as a carbon sequestration booster during the first 30 years of planting. To achieve these results, tōtara must be planted in areas where ungulates are excluded or reduced. Due to the nature of restoration plantings, their spatial distribution and sizes, plantings will achieve higher carbon gains when located on the edge of the forest, avoiding laborious efforts of canopy gap creation (Forbes et al., 2016; Tulod & Norton, 2020). Good management and information-based decisions can achieve higher carbon sequestration for restoration plantings.

Limitations

Only the above-ground carbon pool was measured in this study and below ground, litter and coarse woody debris were excluded. This pool's measurements were only taken once and not repeated over time due to a limited timeframe and restricted permission from landowners and managers. Discrete shrubs were added to the carbon pool by using a cuboid equation. This equation is the best available one but accounts for volume with no carbon as the different shapes of species such as harakeke do not fill a solid cube.

All carbon content values provided the overall carbon content from the year of planting until it was measured. No indication of the amount of carbon sequestered between these time periods and the rate they increase can be obtained from this study.

I was unable to locate any restoration plantings between 31 and 53 years old in the study area and this is reflected in the results, providing a need for more data from plantings 31-53 years old to have information for their carbon content. The lack of available restoration sites in the study area reduced my ability to explore the effect of planting composition of environmental variables on carbon sequestration. This study does not rule out such effects but points out the need for analysis of more planting sites, ideally in a controlled study. Additional environmental variables such as moisture content, rainfall and soil nutrients were not included in this study and might also influence the carbon sequestered.

Survival of the plantings was not accounted for; therefore, results from this study indicate only the CO_2 equivalent carbon amounts successful plantings of tōtara and mixed-species can have in different restoration sites and management in the study area. Areas where plantings failed to survive were covered in gorse, grass and/or bracken. With the use of drone imagery and GIS a specific study that indicated the proportion of failed establishment of plantings and the carbon content in these sites can help better establish a carbon content expectation.

General Conclusion

This is one of the first times that the carbon amounts of restoration plantings established for biodiversity purposes have been calculated. This study showed that restoration plantings in the southern Port Hills and Quail Island have a substantial amount of carbon content that increases with age. These results mean that if we continue to restore and plant our native forest for biodiversity purposes we can gain higher CO_2 equivalent carbon than currently expected. These restored forests already have a high significant biodiversity value and cultural benefits, however now we know that they also contribute more CO_2 equivalent carbon than expected. The CO_2 equivalent carbon amounts of these restoration plantings are higher than those expressed by the MPI look-up tables. These values are also higher to those of regenerating shrublands were Kānuka and/or Mānuka are the dominant species.

Due to the different carbon pools included in the MPI look-up tables the CO_2 equivalent carbon amounts of the restoration plantings in this study are higher up until 30 years from planted. After this the MPI look-up tables show a lag in carbon content until age 50, whereas the CO_2 equivalent carbon amounts of this study show that by 54 to 59 years the CO_2 equivalent carbon is by far higher. This suggests an urgent need to revise the indigenous forest look-up table for it to include values for older plantings and a differentiation between native regenerating shrublands and restoration plantings. These changes will allow managers of future plantings to decide their species selection and numbers around accurate long-term carbon targets. With proper management restoration plantings can continue to sequester carbon and factual carbon credits will provide landowners and managers an extra income that will encourage better and more restoration plantings that provide higher CO_2 equivalent carbon, biodiversity and cultural benefits to our landscapes.

Acknowledgements

I would like to acknowledge the people who have given to this work and my studies without any expectations. I would like them to receive my sincerest gratitude. Firstly, I would like to thank Jeanette Allen for being the first person of the School of Forestry to see potential in me. Thank you to David Norton for being a great supervisor and professor who enlightens his students and has been indispensable for this thesis. His courses and teaching have been an incredible inspiration for my studies and this thesis. Thank you to Euan Mason for being always available and patient, a great teacher and supervisor all through my two years of studies.

I would like to thank the hardest of workers and the most consistent and reliable friend J. Miguel Tapia. He was the best help anyone could have wished to collect data that included tree-hugging, frost pain, being soaked under the rain and receiving free gorse thorns.

Thank you to the disposition of community planting groups and employees, landowners, DOC and council staff of the southern Port Hills and Quail Island. A special thank you for their time and collaboration to Maury Leyland, Max Tweddell, Daniel Aldridge, Ian McLennan, Tony Giles, Murray Lane, Lindsay Daniel, Anne Kennedy, Di Carter, Anthony Shadbolt, Nicholas Head and Laura Molles.

A high appreciation to the scientific leaders that have taken time to address my inquiries. A special thank you to Larry Burrows and Steve Wakelin for their time and help.

To those who helped with the tedious task of reading my work, I thank you, Danielle Thys, Lisa Nguyen, Jarad Sinni, Josh Foster and the Academic Skill Centre staff.

All of my co-workers and classmates, especially Anne Wekesa for making these two years fun and rewarding.

I am grateful for the constant support I received from my family, who is scattered worldwide and especially to my Kiwi family. To my son, who has no idea why Mom talks about trees so much, thank you for sharing my excitement and allowing me to read tree identification books as bedtime stories.

Finally, I would like to thank the T W Adams Scholarship and the Brian Mason Trust's support.

Reference

Asante, P., & Armstrong, G. W. (2012). Optimal forest harvest age considering carbon sequestration in multiple carbon pools: A comparative statics analysis. *Journal of Forest Economics, 18*(2), 145-156. https://doi.org/10.1016/j.jfe.2011.12.002

Beets, P. N., Kimberley, M. O., Oliver, G. R., & Pearce, S. H. (2014). The application of stem analysis methods to estimate carbon sequestration in arboreal shrubs from a single measurement of field plots. *Forests, 5*(5), 919-935. https://doi.org/https://dx.doi.org/10.3390/f5050919

Beets, P. N., Kimberley, M. O., Oliver, G. R., Pearce, S. H., Graham, J. D., & Brandon, A. (2012). Allometric Equations for Estimating Carbon Stocks in Natural Forest in New Zealand. *Forests, 3*(3), 818-839. https://doi.org/https://dx.doi.org/10.3390/f3030818

Beets, P. N., Kimberley, M. O., Paul, T. S. H., Oliver, G. R., Pearce, S. H., & Buswell, J. M. (2014, 1 September 2014). The inventory of carbon stocks in New Zealand's post-1989 natural forest for reporting under the Kyoto Protocol. *Forests, 5*(9), 2230-2252. https://doi.org/https://dx.doi.org/10.3390/f5092230

Bergin, D. O. (2003). *Totara: establishment, growth, and management* (Vol. no. 1). Forest Research.

Bergin, D. O., Kimberley, M. O., & Low, C. B. (2008). Provenance variation in Podocarpus totara (D. Don): Growth, tree form and wood density on a coastal site in the north of the natural range, New Zealand. *Forest Ecology and Management, 255*(5), 1367-1378. https://doi.org/https://doi-org.ezproxy.canterbury.ac.nz/10.1016/j.foreco.2007.10.053

Bickel, K., Richards, G., Köhl, M., Rodrigues, R. L. V., & Stahl, G. (2006). *IPCC Guidelines for National Greenhouse Gas Inventories* (Agriculture, Forestry and Other Land Use), Issue. IPCC.

Bowie, M. H., Madi Corodji, J., Ford, D., Ross, J. G., McLennan, I., Fewster, R. M., & Murphy, E. C. (2018, 15 August 2018). Mouse management on Ōtamahua/Quail Island. *New Zealand Journal of Zoology 45*(3), 267-285. https://doi.org/https://dx.doi.org/10.1080/03014223.2018.1482929

Brockerhoff, E. G., Jactel, H., Parrotta, J. A., Quine, C. P., & Sayer, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity? *Biodiversity and Conservation, 17*(5), 925-951. https://doi.org/https://dx.doi.org/10.1007/s10531-008-9380-x

Bueis, T., Bravo, F., Pando, V., & Turrión, M.-B. (2017). Site factors as predictors for Pinus halepensis Mill. productivity in Spanish plantations. *Annals of Forest Science., 74*(1), 1-10. https://doi.org/https://dx.doi.org/10.1007/s13595-016-0609-7

Burrows, C. J., Wilson, H. D., & Quail Island Restoration, T. (2011). *The island's plants: ecorestoration of Otamahua/Quail Island, 1997-2010*. Ecological Restoration Trust.

Carswell, F., Burrows, L., Easdale, T., Mason, N., Holdaway, R., Payton, I., Fraser, A., Karl, B., & Pearce, G. (2014). Tools to predict carbon sequestration in regenerating shrublands. *Landcare*

Research contract report prepared for the Ministry of Primary Industries. Lincoln: Manaaki Whenua Landcare Research.

Carswell, F. E., Burrows, L. E., Graeme, M. J. H., Norman, W. H. M., & Allen, R. B. (2012). Carbon and plant diversity gain during 200 years of woody succession in lowland New Zealand. *New Zealand Journal of Ecology, 36*(2), 191-202.

Carswell, F. E., Burrows, L. E., Mason, N. W. H., & New Zealand. Department of, C. (2009). *Above-ground carbon sequestration by early-successional woody vegetation: a preliminary analysis* (0478147074;9780478147070). Dept. of Conservation.

Carswell, F. E., Mason, N. W. H., Overton, J. M., Price, R., Burrows, L. E., & Allen, R. B. (2015, January 2015). Restricting new forests to conservation lands severely constrains carbon and biodiversity gains in New Zealand. *Biological Conservation, 181*, 206-218. https://doi.org/10.1016/j.biocon.2014.11.002

Carter, D. (2003). *Towards Tomorrow's Landscapes: Christchurch City Council's Role in Managing Past, Present and Future Habitats on the Port Hills*. https://www.healthyharbour.org.nz/wp-content/uploads/2019/05/Carter-2003-Port-Hills-management.pdf

Carver, T., & Kerr, S. (2017). *Facilitating carbon offsets from native forests*. Motu Economic and Public Policy Research. http://motu-www.motu.org.nz/wpapers/17_01.pdf

Chaplin-Kramer, R., Sharp, R. P., Weil, C., Bennett, E. M., Pascual, U., Arkema, K. K., Brauman, K. A., Bryant, B. P., Guerry, A. D., Haddad, N. M., Hamann, M., Hamel, P., Johnson, J. A., Mandle, L., Pereira, H. M., Polasky, S., Ruckelshaus, M., Shaw, M. R., Silver, J. M., Vogl, A. L., & Daily, G. C. (2019). Global modeling of nature’s contributions to people. *Science, 366*(6462), 255-258. https://doi.org/10.1126/science.aaw3372

Chazdon, R., & Brancalion, P. (2019). Restoring forests as a means to many ends: An urgent need to replenish tree canopy cover calls for holistic approaches. *Science, 365*(6448), 24. https://doi.org/https://dx.doi.org/10.1126/science.aax9539

Clifton, N. C. (1990). *New Zealand timbers : exotic and indigenous : the complete guide.* GP Books.

Commission, C. C. (2021). *Evidence: Chapter 5: Removing carbon from our atmosphere*. Climate Change Comission, New Zealand Government. https://ccc-production-media.s3.ap-southeast-2.amazonaws.com/public/evidence/advice-report-DRAFT-1ST-FEB/Evidence-CH-05-Removing-carbon-from-our-atmosphere-19-Jan-2021.pdf

Coomes, D. A., Allen, R. B., Scott, N. A., Goulding, C., & Beets, P. (2002). Designing systems to monitor carbon stocks in forests and shrublands. *Forest Ecology and Management, 164*(1), 89-108. https://doi.org/https://dx.doi.org/10.1016/S0378-1127(01)00592-8

Coomes, D. A., Flores, O., Holdaway, R., Jucker, T., Lines, E. R., & Vanderwel, M. C. (2014, 27 April 2014). Wood production response to climate change will depend critically on forest composition and structure. *Global Change Biology, 20*(12), 3632-3645. https://doi.org/https://dx.doi.org/10.1111/gcb.12622

Deconchat, M., Brockerhoff, E. G., & Barbaro, L. (2009). Effects of surrounding landscape composition on the conservation value of native and exotic habitats for native forest birds. *Forest Ecology and Management, 258*, S196-S204. https://doi.org/10.1016/j.foreco.2009.08.003

DOC. (2017). *Field protocols for DOC Tier 1 Inventory & Monitoring and LUCAS plots*. Department of Conservation. https://nvs.landcareresearch.co.nz/Resources/FieldManual

Ebbett, R. L. (1998). *The ecology of lowland totara in South Island, New Zealand* [University of Canterbury]. https://ir.canterbury.ac.nz/handle/10092/4833

Ferretti, A. R., & de Britez, R. M. (2006). Ecological restoration, carbon sequestration and biodiversity conservation: The experience of the Society for Wildlife Research and Environmental Education (SPVS) in the Atlantic Rain Forest of Southern Brazil. *Journal for Nature Conservation, 14*(3), 249-259. https://doi.org/10.1016/j.jnc.2006.04.006

Flores, O., & Coomes, D. A. (2011). Estimating the wood density of species for carbon stock assessments. *Methods in Ecology and Evolution, 2*(2), 214-220. https://doi.org/https://dx.doi.org/10.1111/j.2041-210X.2010.00068.x

Forbes, A., Wallace, K., Buckley, H., Case, B., Clarkson, B., & Norton, D. (2020). Restoring mature-phase forest tree species through enrichment planting in New Zealand's lowland landscapes. *New Zealand Journal of Ecology, 44*(1). https://doi.org/10.20417/nzjecol.44.10

Forbes, A. S., Norton, D. A., & Carswell, F. E. (2016). Artificial canopy gaps accelerate restoration within an exotic Pinus radiata plantation. *Restoration Ecology, 24*(3), 336-345. https://doi.org/https://dx.doi.org/10.1111/rec.12313

Ford-Robertson, J. B. (1996). Estimating the net carbon balance of the plantation forest industry in New Zealand. *Biomass & Bioenergy, 10*(1), 7-10. https://doi.org/https://doi-org.ezproxy.canterbury.ac.nz/10.1016/0961-9534(95)00047-X

Genet, R. M. (1997). *The development of a concept plan for the ecological restoration of Quail Island (Otamahua)* [University of Canterbury]. https://ir.canterbury.ac.nz/handle/10092/7065

Genet, R. M., & Burrows, C. J. (1998). *A vision for the ecological restoration of Quail Island (Otamahua)*. Rebus Publications.

Gepp, S., & Wright, M. (2019). Plantation forestry : are the legal and policy settings right to incentivise the right tree in the right place for the right purpose? *Resource Management Journal*, 26-29.

Graeme, M. J. H. (2001, 1 December 2001). Mitigating an Organization's Future Net Carbon Emissions by Native Forest Restoration. *Ecological Applications, 11*(6), 1622-1633. https://doi.org/https://dx.doi.org/10.1890/1051-0761(2001)011[1622:MAOSFN]2.0.CO;2

Griscom, B. W., Adams, J., Ellis, P. W., Houghton, R. A., Lomax, G., Miteva, D. A., Schlesinger, W. H., Shoch, D., Siikamäki, J. V., Smith, P., Woodbury, P., Zganjar, C., Blackman, A., Campari, J., Conant, R. T., Delgado, C., Elias, P., Gopalakrishna, T., Hamsik, M. R., Herrero, M., Kiesecker, J., Landis, E., Laestadius, L., Leavitt, S. M., Minnemeyer, S., Polasky, S., Potapov, P., Putz, F. E., Sanderman, J., Silvius, M., Wollenberg, E., & Fargione, J. (2017). Natural Climate Solutions. *Proceedings of the National Academy of Sciences - PNAS, 114*(44), 11645-11650. https://doi.org/https://dx.doi.org/10.1073/pnas.1710465114

Holdaway, R. J., Easdale, T. A., Carswell, F. E., Richardson, S. J., Peltzer, D. A., Mason, N. W. H., Brandon, A. M., & Coomes, D. A. (2017). Nationally Representative Plot Network Reveals Contrasting Drivers of Net Biomass Change in Secondary and Old-Growth Forests. *Ecosystems, 20*(5), 944-959. https://doi.org/https://dx.doi.org/10.1007/s10021-016-0084-x

Hong, W., Wu, C., & Yan, S. (2004). Modification and its application of generalized Schumacher model. *Ying yong sheng tai xue bao, 15*(2), 241-244.

IPCC. (2003). *Good Practice Guidance for Land Use, Land-Use Change and Forestry.* Intergovernmental Panel on Climate Change (IPCC). https://www.cabdirect.org/cabdirect/abstract/20083162304

Kapfer, J., Hédl, R., Jurasinski, G., Kopecký, M., Schei, F. H., Grytnes, J. A., & Bernhardt-Römermann, M. (2017). Resurveying historical vegetation data – opportunities and challenges. *Applied Vegetation Science, IAVS, 20*(2), 164-171. https://doi.org/https://doi-org.ezproxy.canterbury.ac.nz/10.1111/avsc.12269

Karian, Z. A., & Dudewicz, E. J. (1999). Fitting the generalized lambda distribution to data: a method based on percentiles. *Communications in Statistics - Simulation and Computation, 28*(3), 793-819. https://doi.org/https://dx.doi.org/10.1080/03610919908813579

Kirschbaum, M., Trotter, C., Wakelin, S., Baisden, T., Curtin, D., Dymond, J., Ghani, A., Jones, H., Deurer, M., & Arnold, G. (2009). Carbon stocks and changes in New Zealand's soils and forests, and implications of post-2012 accounting options for land-based emissions offsets and mitigation opportunities. *Landcare Research Contract Report LC0708/174 for MAF, Wellington, New Zealand*.

Leining, C., & Kerr, S. (2018). A guide to the New Zealand emissions trading scheme. *Report prepared by Motu Economic and Public Policy Research for the Ministry for the Environment, Wellington*.

Lu, F., Hu, H., Sun, W., Zhu, J., Liu, G., Zhou, W., Zhang, Q., Shi, P., Liu, X., Wu, X., Zhang, L., Wei, X., Dai, L., Zhang, K., Sun, Y., Xue, S., Zhang, W., Xiong, D., Deng, L., Liu, B., Zhou, L., Zhang, C., Zheng, X., Cao, J., Huang, Y., He, N., Zhou, G., Bai, Y., Xie, Z., Tang, Z., Wu, B., Fang, J., Liu, G., & Yu, G. (2018). Effects of national ecological restoration projects on carbon sequestration in China from 2001 to 2010. *Proceedings of the National Academy of Sciences - PNAS, 115*(16), 4039-4044. https://doi.org/10.1073/pnas.1700294115

Luo, J., Zhang, M., Zhou, X., Chen, J., & Tian, Y. (2018). Tree Height and DBH Growth Model Establishment of Main Tree Species in Wuling Mountain Small Watershed. *Earth and Environmental Science, 108*(4), 42003. https://doi.org/https://dx.doi.org/10.1088/1755-1315/108/4/042003

MAF. (2009). *Look-up tables for the Emissions Trading Scheme*. Ministry of Agriculture and Forestry.

MAF. (2011). *Review of MAF Afforestation Schemes: Permanent Forest Sink Initiative, Afforstation Grant Scheme, East Coast Forestry Project, Sustainable Land Management (Hill Country Erosion) Programme* (Vol. 07). Ministry of Agriculture and Forestry. https://www.mpi.govt.nz/dmsdocument/3729/direct

Mason, N. W. H., Beets, P. N., Payton, I., Burrows, L., Holdaway, R. J., & Carswell, F. E. (2014). Individual-based allometric equations accurately measure carbon storage and sequestration in shrublands. *Forests, 5*(2), 309-324. https://doi.org/https://dx.doi.org/10.3390/f5020309

Milberg, P., Bergstedt, J., Fridman, J., Odell, G., & Westerberg, L. (2008). Observer bias and random variation in vegetation monitoring data. *Journal of Vegetation Science, 19*(5), 633-644. https://doi.org/https://dx.doi.org/10.3170/2008-8-18423

Monge, J. J., Daigneault, A. J., Dowling, L. J., Harrison, D. R., Awatere, S., & Ausseil, A.-G. (2018). Implications of future climatic uncertainty on payments for forest ecosystem services: The case of the East Coast of New Zealand. *Ecosystem Services, 33*, 199-212. https://doi.org/https://dx.doi.org/10.1016/j.ecoser.2018.04.010

MPI. (2017). *A Guide to Carbon Look-up Tables for Forestry in the Emissions Trading Scheme*. Ministry of Primary Industries. https://www.mpi.govt.nz/dmsdocument/4762/direct

MPI. (2018, July 2018). Indigenous Forestry on Private Land: Sustainable Indigenous Forest Management in Accodance with Part 3A of the Forests Act 1949, New Zealand Government

Norton, D. A., Butt, J., & Bergin, D. O. (2018). Upscaling restoration of native biodiversity: A New Zealand perspective. *Ecological Management & Restoration, 19*, 26-35. https://doi.org/https://dx.doi.org/10.1111/emr.12316

Norton, D. A., & Reid, N. (2013). *Nature and farming: sustaining native biodiversity in agricultural landscapes*. CSIRO Publishing.

Oakley, L. (1960). *Harry Ell and his Summit Road: a biography of Henry George Ell*. Caxton Press.

Ogilvie, G. (2009). *The Port Hills of Christchurch* (Fully rev. and updat ed.). Phillips & King.

Omary, A. A. (2011). Effects of aspect and slope position on growth and nutritional status of planted Aleppo pine (Pinus halepensis Mill.) in a degraded land semi-arid areas of Jordan. *New forests, 42*(3), 285-300. https://doi.org/https://dx.doi.org/10.1007/s11056-011-9251-2

Pawson, S. M., Brin, A., Brockerhoff, E. G., Lamb, D., Payn, T. W., Paquette, A., & Parrotta, J. A. (2013). Plantation forests, climate change and biodiversity. *Biodiversity and Conservation, 22*(5), 1203-1227. https://doi.org/https://dx.doi.org/10.1007/s10531-013-0458-8

Payton, I., Barringer, J., Lambie, S., Lynn, I., Forrester, G., & Pinkney, E. (2009). *Carbon sequestration rates for post-1989-compliant indigenous forests*. Landcare Research report LC0809/107 to MAF Policy.

Peters, M. A., Hamilton, D., & Eames, C. (2015). Action on the ground: A review of community environmental groups' restoration objectives, activities and partnerships in New Zealand. *New Zealand Journal of Ecology, 39*(2), 179-189.

Ranney, J. W., Bruner, M. C., & Levenson, J. B. (1981). The importance of edge in the structure and dynamics of forest islands. In *Forest island dynamics in man-dominated landscapes* (pp. 67-95). Burgess, RL & Sharpe.

Reinmann, A. B., & Hutyra, L. R. (2017). Edge effects enhance carbon uptake and its vulnerability to climate change in temperate broadleaf forests. *Proceedings of the National Academy of Sciences, 114*(1), 107-112. https://doi.org/https://dx.doi.org/10.1073%2Fpnas.1612369114

Rüger, N., Condit, R., Dent, D. H., DeWalt, S. J., Hubbell, S. P., Lichstein, J. W., Lopez, O. R., Wirth, C., & Farrior, C. E. (2020). Demographic trade-offs predict tropical forest dynamics. *Science, 368*(6487), 165-168. https://doi.org/10.1126/science.aaz4797

Scion. (2018). *Right tree, right place, right purpose: Scion strategy to 2030* (9780473442651;0473442655;). https://ndhadeliver.natlib.govt.nz/delivery/DeliveryManagerServlet?dps_pid=IE37099995

Shields, M. W., Tompkins, J.-M., Saville, D. J., Meurk, C. D., & Wratten, S. (2016). Potential ecosystem service delivery by endemic plants in New Zealand vineyards: successes and prospects. *PeerJ, 4*(6), 1-22. https://doi.org/https://doi.org/10.7717/peerj.2042

Smith, L. A., Eissenstat, D. M., & Kaye, M. W. (2017). Variability in aboveground carbon driven by slope aspect and curvature in an eastern deciduous forest, USA. *Canadian Journal of Forest Research, 47*(2), 149-158. https://doi.org/https://dx.doi.org/10.1139/cjfr-2016-0147

Steward, G. A., & McKinley, R. B. (2005). Plantation-grown New Zealand kauri: A preliminary study of wood properties. *New Zealand Journal of Forestry Science, 35*(1), 35-49.

Trotter, C., Tate, K., Scott, N., Townsend, J., Wilde, H., Lambie, S., Marden, M., & Pinkney, T. (2005). Afforestation/reforestation of New Zealand marginal pasture lands by indigenous shrublands:

The potential for Kyoto forest sinks. *Annals of Forest Science., 62*(8), 865-871. https://doi.org/https://dx.doi.org/10.1051/forest:2005077

Tulod, A. M., & Norton, D. A. (2020). Regeneration of native woody species following artificial gap formation in an early-successional forest in New Zealand. *Ecological Management & Restoration, 21*(3), 229-236. https://doi.org/https://dx.doi.org/10.1111/emr.12429

Tulod, A. M., Norton, D. A., & Sealey, C. (2019). Canopy manipulation as a tool for restoring mature forest conifers under an early-successional angiosperm canopy. *Restoration Ecology, 27*(1), 31-37. https://doi.org/https://dx.doi.org/10.1111/rec.12707

UN. (2019, 1 March 2019). *General Assembly : Resolution adopted by the General Assembly on 1 March 2019* (69th plenary meeting 1 March 2019), Issue. United Nations. https://undocs.org/A/RES/73/284

Waller, L. P., Allen, W. J., Barratt, B. I. P., Condron, L. M., França, F. M., Hunt, J. E., Koele, N., Orwin, K. H., Steel, G. S., Tylianakis, J. M., Wakelin, S. A., & Dickie, I. A. (2020). Biotic interactions drive ecosystem responses to exotic plant invaders. *Science, 368*(6494), 967.

Wang, V., & Gao, J. (2020). Estimation of carbon stock in urban parks: Biophysical parameters, thresholds, reliability, and sampling load by plant type. *Urban Forestry & Urban Greening, 55*, 126852. https://doi.org/https://dx.doi.org/10.1016/j.ufug.2020.126852

Williams, A. (2010). *On the Ecology and Restoration of Podocarpus cunninghamii in the Eastern South Island High Country. School of Forestry, University of Canterbury* https://ir.canterbury.ac.nz/bitstream/handle/10092/5141/thesis_fulltext.pdf?sequence=1&isAllowed=y

Williams, A., & Norton, D. A. (2012). Estimating carbon stocks in stands of Podocarpus cunninghamii in the eastern South Island high country of New Zealand. *New Zealand Journal of Forestry Science, 42*, 29-38.

Wilson, H. D. (1992). *Banks Ecological Region: Survey report for the New Zealand Protected Natural Areas Programme* (Vol. 21). Department of Conservation.

Xing, D., Bergeron, J. A. C., Solarik, K. A., Tomm, B., Macdonald, S. E., Spence, J. R., & He, F. (2019). Challenges in estimating forest biomass: use of allometric equations for three boreal tree species. *Canadian Journal of Forest Research, 49*(12), 1613-1622. https://doi.org/10.1139/cjfr-2019-0258